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THE USE OF WETLANDS FOR WATER POLLUTION CONTROL

by

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16. ABSTRACT

Wetlands such as marshes, swamps and artificial wetlands, have been shown to remove selected pollutants from urban stormwater runoff and treated municipal wastewaters. Wetlands have produced reduction in BOD, pathogens, and some hydrocarbons, and excel in nitrogen removal. They have been reported to act as sinks for trace metals, phosphorus and suspended solids. Physical pollutant removal mechanisms in wetlands include sedimentation, coagulation, chemical filtration, volatilization, adsorption and chelation. Vegetative mechanisms include absorption through roots, stems and leaves, filtration and chemical transformations in the plants. Chemical transformations of some water borne pollutants also occur in sediment and the water column as a result of anaerobic or aerobic conditions, the presence of catalysts and reactive substances, and with the aid of microbial action. Although individual plant species have been studied for their pollutant removal properties, the interaction of numerous plant and animal species in pollutant removal in a wetland is not well understood.

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FOREWORD

The Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of the environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is that necessary first step in problem solution and it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems for the prevention, treatment, and management of wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, for the preservation and treatment of public drinking water supplies, and to minimize the adverse economic, social, health, and aesthetic effects of pollution. This publication is one of the products of that research; a most vital communications link between the research and the user community.

Application of municipal wastewaters and polluted urban runoff to wetlands has the potential for low-cost water quality protection in numerous communities. The multiple uses of wetlands, such as recreational facilities, wildlife and fisheries enhancement, recharge of ground water, and water quality renovation, offer justification for implementation where single-purpose, energy intensive, conventional treatment facilities may not be supported. The development of appropriate design and operating criteria for wetlands treatment systems is of prime importance to the success of this concept.

Francis T. Mayo
Director
Municipal Environmental
Protection Agency

ABSTRACT

Wetlands such as marshes, swamps and artificial wetlands, have been shown to remove selected pollutants from urban stormwater runoff and treated municipal wastewaters. This report reviews literature on vegetative and hydraulic processes responsible for those removals.

Wetlands have produced reduction in BOD, pathogens, and some hydrocarbons, and excel in nitrogen removal. They have been reported to act as sinks for trace metals, phosphorus and suspended solids.

Physical/chemical pollutant removal mechanisms in wetlands include sedimentation, coagulation, chemical filtration, volatilization, adsorption and chelation. Vegetative mechanisms include absorption through roots, stems and leaves, filtration and chemical transformations in the plants. Chemical transformations of some waterborne pollutants also occur in the sediment layers or the water column as a result of anaerobic or aerobic conditions, the presence of catalysts and reactive substances, and with the aid of microbial action.

Although individual plant species have been studied for their pollutant removal properties, the interaction of numerous plant and animal species in pollutant removal in a wetland is not well understood. Management of wetland vegetative systems, to optimize pollutant removal requires further investigation.

Further research needs to be conducted on long-term impacts to wetlands, bioaccumulation of trace metals, the interaction of individual pollutant removal mechanisms in various wetlands systems, and management techniques for wetlands used as treatment systems.

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SECTION 1

INTRODUCTION

OVERVIEW AND OBJECTIVE

Strong evidence exists to suggest that wetland and upland vegetative systems have the ability to degrade and eliminate various water-borne pollutants. Information on the subject, however, is scattered over a wide range of technical disciplines, much of it the result of investigations undertaken only within the last five to ten years. Consequently, technical guidance is lacking on potentially useful management practices for wetlands. Prior to the adoption of policies for utilizing the natural treatment functions of vegetative systems, there is a need to define the current state of knowledge on the subject. This study was undertaken specifically to address this need. Its objective is to summarize relevant findings on the topic of wetland treatment of pollutants and to guide users to pertinent literature sources for more detailed analysis of specific issues.

BACKGROUND

Over the past decade major steps have been taken to control the discharge of point sources of water pollution. Most apparent is the extensive effort to upgrade and construct wastewater treatment facilities. This has occurred under the Federal Water Pollution Control Act Amendments of 1972 (PL 92-500) and the Clean Water Act Amendments of 1977 (PL 95-217). Also important to improving water quality are efforts to control nonpoint sources of pollution usually associated with stormwater runoff. Numerous studies under the Clean Water Act's Section 208 water quality planning program strongly conclude that water quality goals will not be met in some areas unless specific actions are taken to control these nondiscrete and widespread sources of pollution.

The technology and management tools for controlling surface runoff pollutants have advanced considerably as a result of the 208 program and related research activities. However, for a variety of reasons, principally financial, it appears unlikely that concerted local efforts to implement nonpoint source controls will occur unless these programs can be related to other environmental issues and public needs.

One potentially attractive management option for addressing surface runoff pollution is to tie problem mitigation to a multi-objective framework, including flood control, recreation enhancement, food and fiber production, etc. This concept leads to active consideration of the potential pollutant removal capabilities afforded by wetland and upland vegetated areas. These areas are highly valued for their ecological, agricultural and hydrological significance. Preservation and artificial creation of wetlands is currently the subject of widespread interest because of the recognized importance of these ecosystems as nesting, feeding and nursery areas for birds, fish, invertebrates and wildlife, and as natural runoff detention areas for flood storage and water quality enhancement. Upland pastures and cropland may also offer the means for combining water quality objectives with agricultural open space and landscaping needs.

As stated earlier, the objective of this study is to describe what is and what is not known about the use of wetlands as treatment mechanisms for urban runoff. The report is based upon a survey of scientific investigations and basic literature sources related to nutrient and pollutant cycling in wetland ecosystems. This includes a review of fundamental research regarding plant morphology processes and physical and hydrological relationships, and the growing number of site-specific investigations of pollution impacts and treatment capabilities of vegetative systems. Most investigative work to date has been related to the treatment of municipal wastewater by vegetative systems. This report describes the implications for the evaluation of stormwater runoff treatment. Also incorporated are recent findings from a handful of studies concerned specifically with stormwater runoff, including work by the study team at the Palo Alto Marsh/Flood Basin in California.

REPORT FORMAT

The report is organized into the following nine major sections:

(1) Introduction.

(2) Classification of Practices - Review of definitions and basic characteristics of wetland systems, and description of current methods of using such systems for treatment of wastewaters and stormwater runoff.

(3) Wastewater Characteristics - Identification of basic characteristics of municipal wastewater and stormwater runoff to be considered in the evaluation of vegetative treatment schemes.

(4) Physical and Chemical Pollutant Removal Mechanisms in Wetlands - Description of pollutant removal processes that occur in natural waters and are likely to apply to wetland ecosystems.

(5) Pollutant Removal Processes in Vegetative Systems - Review of waste-processing capacity of vegetative systems, as influenced individually and collectively by characteristic plant types, soil properties and biochemical processes.

(6) Hydrologic Practices - Examination of hydrologic processes and characteristics of wetlands.

(7) Literature and Case Study Review - Summary and comparative review of different types of wetland systems and their influence on water quality and treatment of point and nonpoint source wastewater flows.

(8) Design and Management Guidelines - Summary of wetland design and selection factors common to most situations and specific hydraulic/hydrologic management practices pertaining to pollutant control in wetland systems.

(9) Monitoring and Research Needs - Identification of monitoring needs and research issues, to promote a better understanding of the treatment processes and potential for wetlands use under specific conditions.

SUMMARY

Literature and Practices

Wetland and upland vegetative systems have attracted attention as natural sinks for contaminants and as potential components for wastewater and stormwater treatment systems. The greatest amount of investigation and literature pertains to the treatment of municipal wastewaters. Promising findings have fostered scientific interest and a handful of investigations into the effectiveness of vegetative systems for the control of stormwater pollutants.

Wetlands Treatment of Wastewater

o Wetlands Treatment of Municipal Wastewater

Wetlands occur in a wide range of physical settings at the interface of terrestrial and aquatic ecosystems. Because of this position, some wetlands have been subjected to inadvertent municipal and industrial wastewater discharges for many years. It is only in the past 10 to 15 years that attention has focused on planned use for wastewater treatment. Promising results have been obtained with experimental applications in various natural wetlands including:

o northern peatlands;
o cattail marshes;

- southeastern swamplands;
- cypress domes;
- freshwater/tidal marshes.

Consolidation of results indicates that, in nearly all instances, wetlands act to renovate or improve water quality to some extent. Pollutant removal efficiencies are extremely variable and questions of treatment capacity and long-term wetland impacts are unanswered. Indiscriminant discharge of wastewaters to wetland ecosystems is not advised.

o Artificial Wetlands for Treatment of Municipal Wastewater

The creation of artificial wetlands for treatment of wastewaters has been the subject of experiments for both small and large-scale applications in Europe and the United States, using different types of vegetation and substrates. These systems offer controlled environments for testing and studying vegetative treatment of wastewaters. The resulting data establish hydrologic and constituent balances and assess pollutant removal capabilities for these systems. Examples of artificial systems include:

- meadow-marsh-pond system (New York);
- ponds with reeds or rushes (Germany and Holland);
- peat filters (Minnesota);
- marsh-pond system (California);
- seepage wetland (Michigan);
- water hyacinth ponds (Florida and Texas).

Many researchers favor continued work with artifical wetland systems because of the high degree of control and reliability. The environmental enhancement they provide is an added incentive.

o Wetlands Treatment of Stormwater

Field investigation and research concerning the use of wetlands for treatment of stormwater runoff have been extremely limited. The few studies undertaken (a) exhibit great dissimilarities in the type of wetland and stormwater characteristics examined, (b) contain a very slim data base from which to draw conclusions, and (c) encountered numerous complications in determining hydrologic components. Key investigations include studies of:

- northern peatland (Minnesota);
- cypress wetland (Florida);
- brackish marsh (California);
- high altitude meadows (Lake Tahoe);
- wetland detention basins (Maryland).

In general, these studies revealed the following:

(1) A wide disparity exists in nonpoint source pollution removal capabilities of wetlands, particularly with regard to nutrients;

(2) The greatest consistency in pollutant reduction appears to be for biological oxygen demand (BOD), suspended solids and heavy metals;

(3) The nature of flow and seasonal factors are major influences on pollutant removal capabilities in certain wetlands.

Physical and Chemical Removal Mechanisms in Wetlands

Some investigations of the physical and chemical removal characteristics of wetlands have been undertaken. Several removal processes that occur in natural waters are likely to occur in wetlands. Many have not been documented in marsh environments. Studies of rivers, lakes and oceans have been dominant in research. Nevertheless, a few general conclusions can be drawn:

(1) A wide variety of physical/chemical pollutant removal mechanisms occur in wetlands. Most common are evaporation, sedimentation, adsorption, filtration, chelation, precipitation, decomposition and absorption.

(2) Wetlands exhibit large variations in type, climate and ecosystem. The interaction and relative importance of physical/chemical pollutant removal mechanisms varies significantly among and within wetlands.

(3) Studies of pollutant removal mechanisms in wetlands have generally been piecemeal. Sufficient data have not been collected to formulate a comprehensive theory of pollutant transport and fate within a wetland system.

Vegetative Treatment Systems

Although the initial pollutant removal mechanisms in wetlands are physical and chemical processes, plants can increase the overall capacity of a system to retain or remove pollutants through interactions with various anaerobic and aerobic soil layers, water, and air interfaces. In particular, plant root uptake of pollutants from the sediment frees more exchange sites in the sediments for further pollutant interaction and accumulation. The primary biochemical pollutant uptake and removal processes in vegetative systems are:

(1) Uptake through plant-soil interface, via belowground roots, rhizomes, holdfasts and buried shoots and leaves;

(2) Uptake through plant-water interface, via submerged roots, stems, shoots and leaves;

(3) Translocation through plant vascular system, from roots to stems, shoots, leaves and seeds during growing season;

(4) Differential pollutant uptake, such as preferential storage of trace contaminants in specific plant parts and preferential uptake/accumulation of certain trace elements;

(5) Nonspecific pollutant uptake, occurring primarily as plants absorb large quantities of nutrients from water and sediments;

(6) Uptake and immobilization by plant litter zones, where dead, but not decomposed, plant litter sequesters pollutants through chemical interactions.

- **Nutrient Removal through Wetlands**

Wetland environments present ideal conditions for nutrient cycling and removal, particularly for nitrogen. The aerated water column and aerobic upper sediment layer promote nitrification and the formation of insoluble phosphorus-metal complexes. Reducing (anaerobic) sediment conditions and the interface between the aerobic and anaerobic sediment layers promote ammonification and denitrification.

Wetland vegetation can function as nutrient pumps taking up nitrogen, in the ammonium as well as nitrate form, and phosphorus, in the orthophosphate form. The highest observed nitrogen removal potentials were 300 to 800 kg/ha for aboveground parts of cattails and reeds and up to 1290 kg/ha for belowground parts of rushes and cordgrass. Nitrogen assimilated into wetland vegetation can be translocated back to and stored in the roots during the plant dormancy season or returned to the litter component during senescence of aboveground parts. The highest observed phosphorus removal potentials were 30 to 80 kg/ha for aboveground parts of cattails, reeds and sedges. Phosphorus assimilated into vegetation is not translocated back to the roots, and remains in the plants or plant litter. Hydrologic variables are crucial, particularly during the high runoff season, as particulate matter and organic nitrogen and phosphorus from the litter zone can be flushed. Plant uptake and storage of phosphorus is highly variable and is not a reliable mechanism for phosphorus removal. However, phosphorus-metal interactions can form insoluble complexes which can accumulate into long-term sediment deposits.

- **Uptake and Removal of Trace Elements**

Wetland systems can function as sinks for heavy metals and other trace elements either through vegetative uptake and storage or through immobilization in the sediment layers. The observed heavy metal removal potentials range from:

(1) 0.001 to 0.38 kg/ha of cadmium, with highest removals in *Potamogeton crispus* and *Salicornia pacifica*;

(2) 0.007 to 1.58 kg/ha of copper, with highest levels in *Justicia americana* and *Salicornia pacifica*;

(3) 0.13 to 103.4 kg/ha of iron, with highest levels in *Carex stricta*;

(4) 0.026 to 1.01 kg/ha of lead, with the highest levels in *Salicornia pacifica* and *Phalaris arundinacea*;

(5) 0.001 to 1.714 kg/ha of zinc with the highest levels in *Phragmites communis*, *Carex stricta* and *Scirpus lacustris*.

Although pollutant removal potentials for floating aquatic vegetation are not reported in kg/ha, observed uptake and concentration in water hyacinth (*Eichhornia crassipes*) are significant for some heavy metals, particularly cadmium, chromium, copper, lead, nickel, gold and strontium.

Hydrologic Practices

A clear knowledge of the hydrogeology is crucial for understanding the wetland environment and assessing its potential utility for assimilation of water-borne pollutants. The lack of adequate hydrologic information has hampered the efforts of numerous researchers in their attempt to quantify and evaluate the pollutant removal efficiency of wetlands.

In considering the application of stormwaters and wastewaters to wetlands, the relationships between hydrology and ecosystem characteristics need to be recognized. Factors such as source of water, velocity, flowrate, renewal rate and frequency of inundation have a major bearing on chemical and physical properties of the wetland substrate which in turn influence the character and health of the ecosystem as reflected by (a) species composition and richness, (b) primary productivity, (c) organic deposition and flux, and (d) nutrient cycling. In general, water movement through wetlands tends to have a positive impact on the ecosystem.

Hydrology controls pollutant removal in wetlands through its influence on processes of sedimentation, aeration, biological transformation and soil adsorption. Critical hydrologic factors are:

- o velocity and flowrate;
- o water depth and fluctuation;
- o detention time;
- o circulation and distribution patterns;
- o turbulence and wave action;
- o seasonal and climatic influences;
- o ground-water conditions;
- o soil permeability and ground-water movement.

Various criteria and practices can be identified for hydrologic management of wetlands for improved wastewater and stormwater treatment.

(1) Flow routing - Initial introduction and subsequent distribution of flow should attempt to maximize effective contact between water and wetland soils and vegetation.

(2) Water level maintenance - Manipulation of water levels is a useful means of enhancing pollutant removal by vegetation and soil. Regulation of levels must take into account competing ecosystem needs and the additional nuisance problem of mosquitos.

(3) Inflow/outflow regulation - Possible techniques might include inflow and containment of the "first flush" of runoff, or retention storage during spring runoff until marsh communities are functioning at higher uptake rates.

(4) Seasonal application - Where possible, seasonal applications of wastewaters and stormwaters might be used for specific treatment or flushing purposes, taking into consideration:

  - o biological activity in the wetlands;
  - o availability of dilution flows;
  - o seasonal uses and quality of downstream receiving waters.

(5) Infiltration - Maximum soil contact should be emphasized, with attention given to routing and/or ponding wastewaters in areas of highest soil permeability.

CONCLUSIONS

This study reviews numerous reports of vegetative removal of water-borne pollutants and several studies of wetlands. The conclusions that can be drawn from this review, because of the nature of available literature, reflect only an initial interpretation of the reported literature. Synthesis of these study results into a theory of wetland pollutant removal systems is beyond the scope of this report. Perhaps it is beyond the presently available information. Therefore, conclusions drawn from this literature survey may have to be revised as further research results and operating information are collected.

The conclusions are as follows:

(1) The majority of wetlands studied were able to receive treated municipal wastewater and/or stormwater runoff, remove certain pollutants, and produce satisfactory plant growth. Pollutants removed or decreased included organic wastes (as measured by BOD), nutrients, suspended and volatile solids, and trace metals.

(2) The application of hydraulic controls and vegetation management has the potential to improve wetlands removal of pollutants.

(3) Wetlands remove water-borne pollutants principally through physical and chemical processes that are substantially augmented by biological processes associated with wetland vegetation.

(4) Wetland system stress was reported only in laboratory studies and certain field studies below municipal and industrial discharges, where plants were exposed to excessive pollutant concentrations. Abatement or reduction of pollutant loadings usually led to recovery of wetland vegetation.

(5) Further research should be directed at improving understanding of how wetland systems assimilate pollutants after initial removal.

SECTION 2

CLASSIFICATION OF PRACTICES

The utilization of vegetative systems for the treatment of polluted waters spans a wide variety of practices. This study considers wetland and upland vegetative systems used for the treatment of municipal wastewaters and surface runoff. Included are both natural and artificially developed systems which function in some manner to assimilate water-borne pollutants. In classifying the various practices, attention is given to the underlying purpose(s), basic physical features, principles of operation, and typical areas or instances of application.

DEFINITIONS AND CONCEPTS

Prior to examining specific practices, it is important to clarify what constitutes wetlands and uplands and to distinguish between natural and artificial vegetative systems.

Wetland Characteristics

The term "wetlands" is a relatively new designation for marshes, bogs, swamps and other low-lying land dominated by saturated soil conditions [2-1]. Numerous efforts have been made to define wetlands in terms of function and characteristics. Most definitions reflect the fundamental role of water and hydrogeological factors in wetlands [2-2]. Cowardin et al. [2-1] have established the following definition to be used in conjunction with the ongoing national wetlands inventory.

> "Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For purposes of this classification wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year."

o Natural Wetlands

Descriptions of the principal types of natural wetlands, according to hydrological factors, are provided in Table 1. Included are (a) riverine, (b) lacustrine, (c) palustrine, (d) brackish, and (e) tidal. The system characteristics, representative vegetation types and some of the important environmental sensitivities are also shown. In this classification system, based on Cowardin et al. [2-1], saline tidal wetlands are represented by the estuarine system.

In general, systems with open stands of emergent vegetation and/or large open areas allowing algal growth demonstrate the highest plant productivity. Palustrine freshwater marshes with peat-filled bogs or heavily shaded wooded swamps generally do not show as high a productivity as other fresh and salt water marshes. Lacustrine marshes are strongly regulated by the water level in the system. Emergent plants are limited to pond and lake fringes and generally do not occur where the water depth exceeds 1 to 2 meters. Floating and submerged vegetation typically occur in the upper water layers (1 to 2 meters depth) where thick plant growth can reduce dissolved oxygen and attenuate light, discouraging plant growth in the lower layers.

Brackish marshes often occur within salt marsh areas near streams, and in many cases can be considered as a type of salt marsh. In Table 1, brackish marshes are identified as a separate category to represent marshes subject to salinity changes but not influenced by tidal actions, as in the case of salt pans and seasonal inland marshes.

o Artificial Wetlands

Artificial wetlands are the result of establishing wetland vegetation and the required hydrologic conditions in locations where they previously did not exist. Examples might include the creation of ponds and marshes for wildlife and aquatic habitat enhancement. Lowlands converted for use as permanent stormwater retention basins are another possibility. Trenches and ponds constructed specifically for physical and biological treatment of wastewaters may also take on the form of artificial wetlands. In many such cases, the substrate and vegetation established are foreign to the immediate locale. Table 2 provides a listing of some of the principal types of artificial wetlands encountered in this study.

All of these systems exhibit some of the basic ecologic features presented earlier in Table 1 under natural wetland systems. Artificial marsh systems function in a similar manner to riverine systems where there is ample circulation, an abundance of wetland vegetation, and maximum interaction between water, vegetation, and sediments.

TABLE 1. CLASSIFICATION OF NATURAL WETLAND SYSTEMS[a]

Wetland system	System characteristics	Vegetation types	Sensitivities
Freshwater Marshes - Riverine (associated with water channels)	Water circulation distributes dissolved and suspended materials through system. Good aeration and light penetration.	Emergent plants - cattails, reeds, sedges, bulrush, watercress; floating algae.	Subject to sedimentation, scouring and seasonally-changing water levels. Pollutant loadings vary with watershed.
Freshwater Marshes - Lacustrine (associated with ponds and lakes)	Temperature/oxygen stratification and light attenuation can cause major differencees in top, middle and bottom layers. Circulation is often poor.	Floating plants - duckweed, water fern, water primrose, pondweeds and others. Emergent plants - see riverine system; submerged plants.	Closed or semi-closed systems. Pollutants enter food chain or accumulate in sediments.
Freshwater Marshes - Palustrine (not confined by channels or adjacent to lakes)	Surface layer has thick and/or porous deposits with high organic content. Marsh is fed by subsurface seepage/high ground water.	Peat bogs, cypress, mangrove and papyrus swamps - vegetation types often specific to geographical area.	Isolation from open water bodies (streams, rivers and lakes) limits water exchange, forming potential pollutant sink.
Brackish Marshes - (salinity > 0.4 ppt[b])	Marsh fed by seasonal surface flows and/or seepage; may also be subject to tidal influences; can experience salinity fluctuations.	Emergent plants - sedges, bulrush, pickleweed, saltgrass, saltbush.	Evaporation can lead to salinities of 60-80 ppt and concentration of pollutants.
Salt Marshes - (subject to tidal influence)	(1) wetlands near streams (2) lower wetlands - reversing flow (3) lower wetlands - drained only at low tides (4) upper wetlands - inundated only at high tides	Emergent plants - pickleweed, cordgrass, sedges, saltgrass.	Salinity and sediment interactions can trap pollutants; however, low pH and oxidizing muds can re-release pollutants to system on a continuing basis.

a. Derived from references 2-2, 2-3 and 2-4.

b. ppt, parts per thousand

TABLE 2. ARTIFICIAL WETLANDS USED FOR THE TREATMENT OF WASTEWATER OR STORMWATER[a]

Type	Description
Marsh	Areas with impervious to semi-pervious bottoms planted with various wetlands plants such as reeds or rushes.
Marsh-pond	Marsh wetlands followed by pond.
Pond	Ponds with semi-pervious bottoms with embankments to contain or channel the applied water. Often, emergent wetland plants will be planted in clumps or mounds to form small subecosystems.
Seepage wetlands	Wastewater irrigated fields overgrown with volunteer emergent wetland vegetation as a result of intermittent ponding and seepage of wastewater.
Trench	Trenches or ditches planted with reeds or rushes. In some cases, the trenches have been filled with peat.
Trench (lined)	Trenches lined with an impervious membrane usually filled with gravel or sand and planted with reeds.

a. Derived in part from reference 2-2.

Pond systems function similarly to a freshwater lacustrine system. The type of treatment depends on the design of the basin. Deep basins without cultivated vegetation function as simple wastewater treatment lagoons where sedimentation, algal growth and adsorption to particles occur. With aeration, growth of microorganisms is promoted and breakdown of some organic material occurs. The same processes occur to a lesser degree without aeration, due to the presence of different types of microorganisms. With the growth of submerged and floating plants such as duckweed and water hyacinth, contaminants can be absorbed through plant metabolism. Primary productivity and sedimentation can occur at a high rate in these systems, which leads to pollutant sinks forming in the vegetation and sediments. Since the water level in these systems can be controlled, the vegetation can be skimmed from the surface and harvested, and the bottom deposits can be scraped or dredged out.

Seepage wetlands, and lined and unlined trenches fall roughly in the category of palustrine systems. Water applied to these systems generally does not exit as surface runoff but infiltrates through the wetland layers to the ground water or to underdrains. Vegetation may aid in the treatment process, but the most critical factor is the use of soils with high permeability rates, such as gravelly soils and alluvial deposits. The success of an infiltration system often depends on alternate wet and dry cycles to allow for proper soil reaeration. In this sense, it probably does not function as a true wetland system and can best be described as a hybrid wetland-dryland system.

Upland Characteristics

In a technical sense, uplands are commonly thought of as those areas elevated above river valleys and plains. For the purpose of this study a broader view is taken. Upland vegetative systems are considered to encompass all areas not falling within the definition of wetlands, i.e., where the water table is not near the surface during most of the year and where conditions are not favorable for the formation of hydric soils and the support of hydrophytes. The upland category is intended to include rangelands, croplands, pastures and forested lands. Such areas are subjected to surface runoff and perhaps inundation for extended periods. Upland areas are also used extensively for land application of wastewater. The soils and vegetation are not, however, the result of, nor dependent upon, excessive soil moisture or standing water. Grassed stormwater detention basins and grassed drainage swales, which serve to pass intermittent runoff flows, may also be considered upland vegetative systems.

The soil column plays an important role in the function of an upland system. In the removal of contaminants, it acts as a site for physical interactions such as filtration and adsorption, chemical interactions such as precipitation, adsorption and formation of organic complexes, and biological interactions such as bacterial metabolism, plant metabolism and absorption. Chief factors governing the movement of pollutants in these interactions are the soil pH, oxidation-reduction potential (redox potential) and the mineral composition of the soil. High pH and low redox potential lead to the formation of insoluble metal complexes, which effectively prevent plant uptake of metals. As pH decreases and the redox potential increases (a situation that can occur in temporarily flooded soils), metal ions are released and available for plant uptake. In alkaline soils, heavy metals can be immobilized as precipitates. Soil reaeration is important for the maintenance of physical, chemical and biological interactions. The cation exchange capacity (CEC) governs the long-term accumulation of pollutants such as heavy metals. For the protection of ground-water resources, particularly where used for human or livestock supply, the maximum limit of pollutant accumulation should be calculated for each soil type.

VEGETATIVE SYSTEMS FOR WASTEWATER TREATMENT

Although inadvertent application has occurred for many years in some instances, the planned utilization of vegetative systems for treatment of municipal wastewater is a relatively recent innovation. Described here are the three principal categories of interest: (a) natural wetlands, (b) artificial wetlands, and (c) uplands.

Natural Wetland Treatment of Wastewater

The interest in utilizing natural wetlands for treatment of wastewater effluent has emerged as a result of several factors:

(1) Public demands for more stringent wastewater effluent standards;

(2) Rapidly escalating energy costs associated with conventional treatment facilities;

(3) Recognition of the natural treatment functions readily available in wetland ecosystems;

(4) Appreciation of the aesthetic, wildlife and other incidental environmental benefits associated with wetlands preservation and enhancement.

In most cases to date, the primary objective has been to provide a degree of tertiary treatment, mainly to meet nutrient removal requirements. Discharge of secondary effluent to wetlands seeks to take advantage of natural treatment processes related to four principal features of wetlands [2-1]:

(1) Dispersion of surface water over a large area through intricate channelization of flow;

(2) Utilization and transformation of elements by microorganisms;

(3) Physical entrapment through sorption in the surface soils and organic litter;

(4) Uptake and metabolic utilization by plants.

Achievement of water quality treatment standards through the use of natural biological systems is found to offer an economically attractive alternative to conventional advanced wastewater treatment practices [2-2, 2-5, 2-6].

Possible constraints or problems associated with the use of wetlands for wastewater treatment include the following:

(1) Nutrient and organic loading capacities are unique to individual wetlands [2-7];

(2) Proximity to sources of wastewater effluent influences cost [2-5];

(3) Long-term ecosystem effects are largely unknown [2-2, 2-7];

(4) Wetlands may provide breeding grounds for disease and insects, and may generate odors [2-2].

Experimentation with wastewater discharges to natural wetlands has occurred in many different parts of the U.S. and Canada. Examples of wetlands that have been examined for wastewater treatment include the following:

- cypress domes (Florida) [2-9, 2-10, 2-11];
- northern peatlands (Michigan, Wisconsin) [2-12, 2-13];
- cattail marshes (Wisconsin) [2-14, 2-15, 2-16];
- freshwater/tidal marsh (New Jersey) [2-17, 2-18];
- lacustrine marsh (Canada) [2-19];
- swamplands (Canada) [2-8];
- southern freshwater marshes (Florida) [2-20, 2-21, 2-22].

As is evident from this listing, a range of different types of wetlands are under investigation. Some of the wetland systems have been receiving wastewater discharges for long periods of time while others are the subjects of recently initiated pilot studies.

Artificial Wetland Treatment of Wastewater

The use of artificial wetlands for wastewater treatment seeks to take advantage of many of the same principles that apply in natural biological systems, but does so within a more controlled environment.

Small-scale wetlands have been created expressly for the purpose of providing wastewater treatment [2-23, 2-24, 2-25], while others on a larger scale have been implemented with multi-use objectives in mind, i.e., using treated sewage effluent as a freshwater source for the creation and restoration of marshes for environmental enhancement [2-26, 2-27].

The use of artificial wetlands for wastewater treatment is founded largely on the work of Kathe Seidel and her co-workers at the Max Planck Institute in Germany [2-23]. Since the early 1950's, they have been studying the wastewater treatment capabilities of plants established on artificial substrates. More recent developments have included the use of peat filters, floating plants (e.g., duckweed and water hyacinths), and large-scale creation of wetland habitats. Artificial wetlands have been studied for their capabilities to provide primary and secondary wastewater treatment as well as for advanced or tertiary treatment. They afford much greater operational flexibility than do natural systems. Some systems are set up to recycle a portion of the wasteflow and to direct the final effluent into the soil for recharge purposes [2-25]. Others act as flow-through systems, discharging final effluent to receiving waters [2-26, 2-27].

Possible constraints to the use of artificial wetlands for wastewater treatment include:

(1) Geographical limitations of plant species. There is also the potential that a plant species introduced into new areas will become a nuisance or an agricultural competitor;

(2) Land-based treatment systems require 4 to 10 times more land area than a conventional wastewater treatment facility;

(3) Plant biomass harvesting is constrained by high plant moisture content and wetland configuration;

(4) Wetlands may provide breeding grounds for disease and insects, and may generate odors.

Examples of typical artificial wetland systems used for wastewater treatment include the following:

- meadow-marsh-pond system (New York) [2-25];
- ponds with reeds or rushes (Germany, Holland) [2-23, 2-24];
- peat-filled trench systems (Finland) [2-28];
- peat filters (Minnesota) [2-29];
- marsh-pond system (California) [2-26];
- seepage wetland (Michigan) [2-27];
- hyacinth pond systems (Texas, Florida) [2-30, 2-31, 2-32].

These examples show a wide geographical distribution and active interest in a range of wetland vegetative systems. There are currently many in the field promoting expanded utilization of artificial, rather than natural, wetland systems [2-7, 2-14].

Upland Treatment of Wastewater

Major efforts have been made during the past several years to advance the practice of utilizing upland soil and vegetation for the treatment and disposal of various types of wastewater [2-33]. In the U.S., the use of land treatment for municipal wastewater has been increasing steadily over the past 40 years. The increase is attributable to:

(1) Public demands and policies calling for higher levels of waste treatment and reduction of surface water discharges;

(2) Recognition of the resource value of wastewater, particularly in the western states.

Land treatment is a versatile practice, in which a variety of processes may be used to achieve a number of different objectives. Specific purposes for which land treatment of wastewater has been employed include:

- water quality protection;
- wastewater reclamation and reuse;
- ground-water recharge;
- nutrient recycling;
- crop production.

The principal methods of land treatment in common use are (a) slow-rate infiltration, (b) rapid infiltration, and (c) overland flow. Illustration of these methods is provided in Figure 1.

o Slow-Rate Infiltration

The slow-rate infiltration system is normally an integral component of agricultural or landscaping operations. Vegetation is a critical factor for managing water and nutrients in such systems. Slow-rate infiltration relies upon soil, plants and microorganisms to renovate wastewater as it passes through the soil profile.

The portion of applied wastewater that is not consumed through evapotranspiration percolates to the ground water. Typically, slow-rate ground-water systems are designed to preclude any surface runoff of applied wastewater. Sprinkle and surface application techniques are both commonly used in slow-rate land treatment systems.

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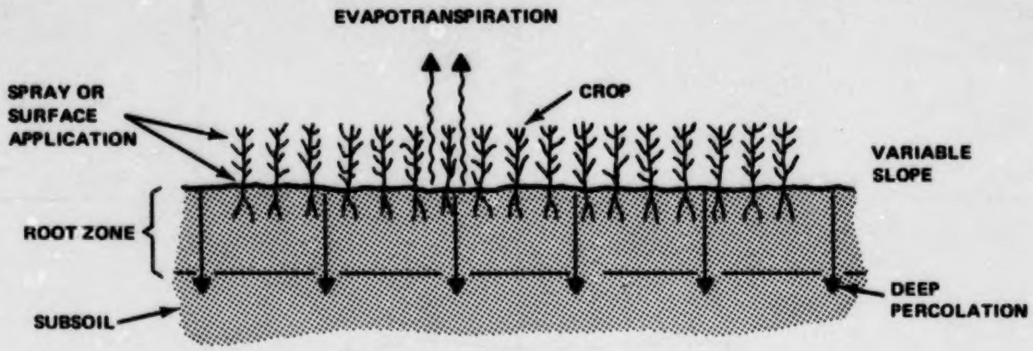
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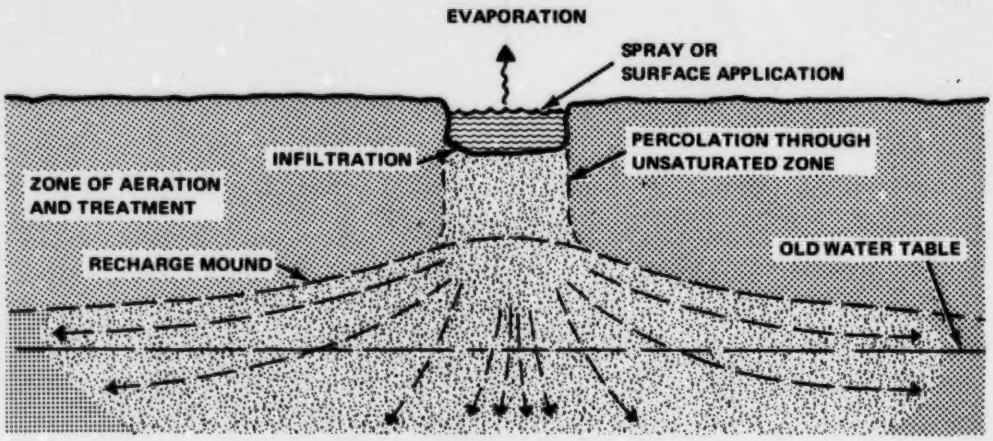
SLOW-RATE INFILTRATION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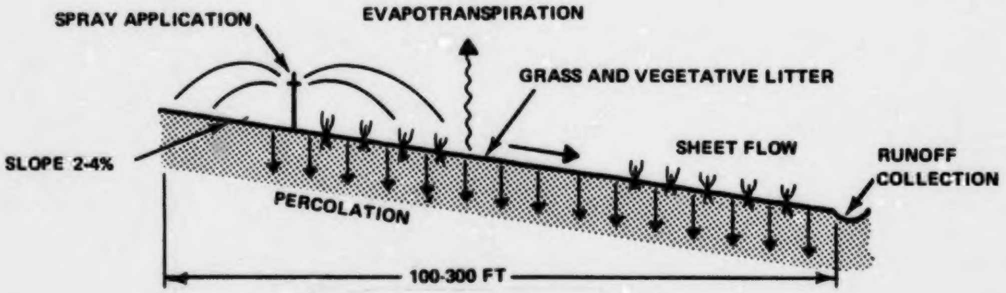
RAPID-RATE INFILTRATION


SPRAY APPLICATION
EVAPOTRANSPIRATION
GRASS AND VEGETATIVE LITTER
SHEET FLOW
RUNOFF COLLECTION
SLOPE 2-4%
PERCOLATION
100-300 FT

OVERLAND FLOW

Figure 1. Methods of land application.[a]

a. Source: reference 2-23.

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Geographically, slow-rate systems may be found in use in all climatic regions of the U.S. The major distinctions are in the period and rates of application and resultant storage requirements. Controlling factors are soil conditions, type of crop or vegetation, and climate.

o Rapid Infiltration

In contrast to slow-rate systems, rapid infiltration is usually employed independently of agricultural or landscaping operations. Wastewater is applied to thick deposits of coarse, permeable soils, where it readily percolates through the soil, eventually reaching underlying ground water. Although there are exceptions, vegetation is typically not used. Physical, chemical and biological treatment of wastewater occurs as a result of passage through the soil matrix.

Application of wastewater is accomplished by spreading in basins or by sprinkling. Recovery of the percolated effluent through underdrains or well fields is often incorporated into such systems to achieve reuse objectives or to protect the quality of particularly sensitive ground-water bodies.

The use of rapid infiltration systems is widespread in the U.S. The primary criteria dictating feasibility are soil and hydrogeological compatibility.

o Overland Flow

Overland flow is a method of land treatment used on relatively impermeable soils. In such systems, wastewater is typically applied at the top of gently sloping terrain and allowed to flow across the vegetated surface until it is collected in runoff ditches some distance downslope. The collected runoff is then available for reuse or direct discharge to surface waters.

Overland flow (also referred to as grass filtration) mainly relies upon microbial and plant activity in surface soils and a dense grass cover for renovation of wastewater. It has been used for secondary treatment as well as to achieve high levels of nitrogen and BOD removal, comparable to conventional advanced wastewater treatment processes.

Overland flow is a relatively new practice in the U.S., although such systems have been in operation for several decades in Australia. The most promising results have been with systems in warm climates. Recent work has been undertaken to develop information concerning performance and suitability of overland flow systems in cold regions.

VEGETATIVE SYSTEMS FOR STORMWATER TREATMENT

The primary objectives of stormwater management are to minimize inconvenience, hazards and damage resulting from the accumulation of stormwater runoff. The conventional approach has been to provide a system of channels and pipes to carry away rainfall as quickly and efficiently as possible [2-34]. Because of secondary hydrologic impacts and nonpoint source water quality concerns, traditional stormwater drainage practices have been subjected to serious reevaluation [2-35].

A major new emphasis is being placed upon the identification and application of "natural" engineering techniques to preserve and enhance the natural features of a site, and to maximize economic-environmental benefit [2-34]. Natural engineering techniques are those which capitalize on, and are compatible with, natural resources and processes [2-34]. Foremost are the uses of wetland and upland vegetated areas for detention, infiltration and treatment of runoff.

Wetlands Treatment of Stormwater

By nature, most wetland systems receive surface runoff from adjacent lands and watercourses. To varying degrees, this provides treatment of runoff waters. In the past few years serious attention has been given to capitalizing on wetland processes as a means of providing detention storage and treatment of stormwater flows. The emerging interest may be attributable to three factors:

(1) Increased knowledge and concern regarding the control of nonpoint source stormwater pollutants;

(2) Heightened interest and experience with natural biological treatment systems for wastewater pollutants;

(3) Alarmed concern for preservation of this nation's diminishing wetland resources.

Many wetlands have been receiving inadvertent discharges of stormwaters for a number of years. To date, there have been only a few instances where stormwater has been specifically routed into natural or artificially-created wetlands for flood control or water quality management purposes. Where the practice has been employed, consistent reduction of BOD, suspended solids and heavy metals generally have been shown. Stormwater treatment through wetlands encompasses three categories:

(1) Systems planned primarily for flood control with treatment as an incidental benefit;

(2) Systems planned and operated with treatment of stormwater pollutants as a primary objective;

(3) Existing wetland systems providing detention and treatment of stormwater flows as an unplanned, natural function.

The factors responsible for wetlands treatment of surface runoff are largely the same as those noted previously in reference to wastewater treatment. In principle, wetlands offer hydraulic resistance to surface runoff flowing through them, resulting in decreased velocities and increased deposition of suspended sediments. The large surface area provided by surface soils and vegetation contributes to higher levels of absorption, adsorption, microbial transformation and biological utilization than normally occur in more channelized water courses [2-13].

The utilization or creation of wetlands for stormwater treatment is limited by several factors, including:

(1) Proximity of natural wetlands to sources of runoff needing treatment;

(2) Seasonal and sporadic nature of stormwater runoff where reliable water supply is needed for maintenance of wetland vegetation;

(3) High flows and flushing action associated with runoff events;

(4) Potential for creating nuisance vector and odor problems.

A variety of management practices have been suggested to overcome these limitations and enhance the overall capabilities of wetlands for treatment of stormwater runoff. A detailed discussion of possible measures, including techniques related to inflow/outflow regulation, water level manipulation and flow distribution, is provided in Section 6.

Examples of instances where wetlands have been employed, examined or proposed for treatment of stormwater include:

- northern peatlands (Minnesota) [2-36];
- cypress wetland (Florida) [2-37];
- brackish marsh (California) [2-38];
- high altitude meadows (California) [2-39];
- vegetated retention basins (Maryland) [2-40];
- southern freshwater marsh (Florida) [2-41].

Upland Retention and Treatment of Stormwater

During the past 10 to 15 years, one of the main focuses of stormwater management has been on methods of retarding and attenuating stormwater runoff in urban settings and other upland areas. The purpose has been to provide acceptable levels of drainage relief and flood control. A number of secondary functions are also served, including [2-42]:

- ground-water recharge;
- recreation and open space;
- surface water pollutant reduction;
- erosion control.

Three categories of practices which utilize the upland landscape for surface runoff control can be identified:

(1) Detention/retention storage basins;
(2) Natural overland flow;
(3) Vegetated/grassed waterways.

o Detention/Retention Storage Basins

The use of landscaped basins for the detention or retention of surface runoff has become a popular stormwater management technique throughout the U.S. during the past 10 years. Such facilities may take the form of either onsite or offsite impoundments where runoff is collected for (a) controlled downstream release (detention), or (b) containment and infiltration (retention), or a combination thereof. The design of such facilities will vary depending upon the land costs, space availability, physical and aesthetic characteristics of the area, topography, climate and other local factors [2-42]. Whether or not the detention/retention facility is to serve multipurpose uses, such as runoff treatment/disposal and recreation, is a factor that may dictate size, shape, depth and landscaping.

Detention/retention storage basins may provide for surface runoff pollution control in a variety of ways [2-43]:

(1) Infiltration and entrapment of pollutants in the soil;

(2) Flow detention and velocity reduction which minimizes erosive forces in downstream areas;

(3) Delay and modulation of runoff pulses and consequent shock loading of pollutants to receiving waters;

(4) On-site storage reduces flood-related surface water flows across urban street surfaces where large quantities of pollutants may be picked up and transported by runoff;

(5) Reduction of stormwater pollutant loadings, through the settling of particulate matter and biological oxidation of organic materials in ponding areas.

Vegetation at detention/retention facilities, primarily, is planted for aesthetic reasons; however, other specific purposes might include [2-42]:

- recreational needs (i.e., turfed ballfields);
- improved infiltration;
- erosion control;
- barriers to auto and pedestrian access.

The planned use of vegetation for uptake or reduction of surface runoff pollutants in these facilities has gained little attention to date.

o Natural Overland Flow

The technique of causing surface runoff to follow natural overland sheet flow routes has become a preferred method of handling residential storm drainage when practicable [2-34]. The recommended practice is to direct the runoff flow over and through turf, or other flow retardants such as ground cover or forest litter. The longest overland flow distances possible are desired.

Stormwater treatment functions served by natural overland flow are similar to those achieved through detention/retention storage. In particular this practice provides for:

- velocity and scour control;
- flow detention and lengthening of the time of concentration;
- retention and deposition of pollutants;
- some infiltration of stormwater.

The main application of overland flow has been in the planning and design of newly developing residential areas, particularly where emphasis on maintenance of natural surroundings has a high priority. Examples can be found throughout the U.S. Foremost, perhaps, is the Woodlands Development in Houston, Texas, where overland flow of surface runoff constitutes one of a number of natural drainage techniques aimed at minimizing the hydrologic and water quality impacts associated with urbanization [2-44]. In this instance, maintenance of natural terrain and ground cover were important criteria. In other cases, preliminary site grading and artificial landscaping is done in order to establish an overland flow drainage system.

The idea of treating large volumes of collected surface runoff by overland flow is currently under investigation in Utah. Under the National Urban Runoff Program, an overland flow treatment strategy has been devised which will involve spreading surface runoff over a pasture area [2-45]. This is intended to test the effectiveness of a natural grassland in removing stormwater pollutants. The system design will rely largely on concepts borrowed from overland flow practices for sewage treatment.

o **Grassed Waterways**

Grassed or vegetated waterways have been used for many years in connection with agricultural operations. Their primary purpose is to convey surface runoff from a field, diversion or terrace with minimal soil loss and flooding. The stabilization provides erosion control in the channel by reducing the velocity of water near the bed [2-46, 2-47]. Additionally, the decrease in water velocity allows for greater infiltration of surface water and increased deposition of sediment. The potential use of grassed waterways for removal of other stormwater pollutants has recently come under investigation [2-48, 2-49]. Important considerations and constraints regarding the use of grassed waterways are:

- velocity and capacity of the water channel;
- large land area requirements;
- soil types;
- ground-water conditions;
- vegetative tolerance to drought and inundation.

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SECTION 3

WASTEWATER CHARACTERISTICS

There is no known limitation on the quality of municipal wastewater or urban runoff that may be successfully treated by vegetative practices. In practice, treated wastewater and runoff representative of typical conditions have been applied to vegetative systems. It is likely that "typical" waters, particularly those at the cleaner end of the range, would predominate in near-future applications. This section identifies basic characteristics of municipal wastewater and runoff, permitting a potential user of a vegetative treatment scheme to compare the quality of the proposed waste with "typical" and with case study waters discussed in subsequent sections.

MUNICIPAL WASTEWATER CHARACTERISTICS

Raw Sewage

Untreated municipal wastewater has not been consciously applied directly to marshes or other vegetative systems, nor have studies of the effect of these systems upon raw sewage quality been uncovered. However, the quality of treated municipal effluents depends upon the nature of the raw sewage and the selection of unit treatment processes to remove specific pollutants from the sewage. Typical pollutant concentration ranges for raw sewage are presented in Table 3.

Treated Wastewater

The most commonly encountered wastewater treatment stages are (a) primary, where sedimentation removes the bulk of settleable solids and a portion of organic matter, and, (b) secondary, where biological oxidation produces significant reduction in the balance of the organic matter. Variations of the two basic treatment stages include chemical coagulation, extended aeration, filtration, algal polishing, chemical destruction (e.g., chlorination) and air stripping. These variations can be part of the basic processes of sedimentation and oxidation or can be added to a process train. Each variation or additional unit process is generally targeted at a specific group of wastewater constituents such as suspended particulates, colloids, organic matter, nutrients, toxics, pathogens and dissolved salts. Most additional unit treatment processes are designed based on an influent of typical primary or secondary effluent (Table 4). As a starting assumption, primary or secondary effluent would be the most common wastewater supplied to

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TABLE 3. COMPOSITION OF RAW MUNICIPAL WASTEWATERS

Constituent	Raw wastewater range, mg/l Metcalf & Eddy, Inc.[a]	 U.S. EPA[b]
Solids, total	350-1,200	700-1,000
Dissolved solids, total	250-850	400-700
Mineral	145-525	250-450
Volatile	105-325	150-250
Suspended solids, total	100-350	180-300
Mineral	30-75	40-70
Volatile	70-275	140-230
Settleable solids, (ml/liter)	5-20	--
Settleable solids, total	--	150-180
Mineral	--	40-50
Volatile	--	110-130
Biochemical oxygen demand, (BOD_5) $20^{o}C$	100-300	160-280
Ultimate carbonaceous	--	240-420
Ultimate nitrogenous	--	80-140
Total oxygen demand (TOD)	--	400-500
Chemical oxygen demand (COD)	250-1,000	550-700
Total organic carbon (TOC)	100-300	200-250
Nitrogen, total as N	20-85	40-50
Organic	8-35	15-20
Ammonia	12-50	25-30
Nitrites	0	0
Nitrates	0	0
Phosphorus, total as P	6-20	10-15
Organic	2-5	3-4
Inorganic	4-15	7-11
Chlorides added	30-100	50-60
Alkalinity added, as $CaCO_3$	50-200	100-125
Grease	50-150	90-110

a. Source: reference 3-1.
b. Source: reference 3-2.

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TABLE 4. COMPOSITION OF TREATED MUNICIPAL WASTEWATERS

Constituent	Primary effluents, mg/l			Secondary effluents, mg/l		
	Average[a]	Range[a]	Range[b]	Average[a]	Range[a]	Range[c]
BOD_5	136	70-200	68-243	25	15-45	4.4-241
COD	330	165-500	--	55	25-80	--
Suspended solids	80	40-120	27-100	15	10-30	19-129
Nitrogen, total as N	35	--	16-45	30	--	23-48
Ammonia, as N	--	--	9-27	--	--	5-29
Phosphorus, total as P	7.5	--	1.2-9.8	5.0	--	1.2-9.1
Total coliforms, MPN/100 ml	$2x10^6$	$5x10^6$-$5x10^8$	--	$1x10^3$	$1x10^2$-$1x10^4$	--

a. Source: reference 3-6.

b. Derived from 1978 end-of-year-average self monitoring reports for San Francisco (3 plants), Pittsburg, East Contra Costa, San Mateo, Benicia, Pacifica, Estero and Sausalito, California. Unpublished NPDES data.

c. Derived from 1978 end-of-year-average self monitoring reports for Central Contra Costa, East Bay Municipal Utilities, San Jose, Redwood City, San Carlos-Belmont, Hayward, North San Mateo Co., Oro Loma, Novato-Ignacio, Petaluma, Sonoma Valley and Mountain View, California. Includes activated sludge, trickling filters and lagoons. Unpublished NPDES data.

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vegetative treatment systems. Additional treatment processes, added before discharge of effluent to a marsh or wetland, would usually reduce concentrations of certain wastewater constituents, such as toxic metals, or change their chemical nature, such as converting ammonium to nitrate.

Trace Elements in Municipal Wastewaters

Of particular concern in the planning of a vegetative system for the treatment of municipal wastewater effluents is the presence of trace elements, often referred to collectively as trace metals. These elements include copper, lead, silver, cadmium, chromium, arsenic, mercury and zinc. All these elements are of interest because of their potential, when present in sufficient quantities, for toxicity to living organisms, including those responsible for biological activity in a wastewater treatment process. The presence of trace elements in municipal wastewater can vary by orders of magnitude, chiefly depending upon the addition of industrial wastes to the municipal sewage system. Trace elements can be present in natural water supplies, added from materials used in water distribution systems, or from purely domestic wastes. Table 5 presents reported trace elements in raw municipal wastewaters. Table 6 lists trace elements in treated wastewaters. Since the sources of the data are generally not the same for raw and treated wastewaters, the values presented are not precisely indicative of the effectiveness of levels of treatment and should only be used as typical concentrations in different types of effluent.

Wastewater Particle Sizes

To the relative extent that a vegetative treatment system would employ sedimentation, filtration, adsorption and absorption as pollutant removal mechanisms, the distribution of pollutants in a particular waste would affect the potential efficiency of the treatment system. Figure 2 and Table 7 present size distributions of wastewater particles. It should be noted that with increasing treatment of wastewater from raw sewage to secondary effluent, there are greater reductions in colloidal and settleable material than in soluble or supra-colloidal matter. Thus, with increasing effluent quality discharged to a vegetative treatment system, the principal pollutant removal mechanisms in the vegetative system should shift from sedimentation or filtration to adsorption and absorption, and the system may have to be selected accordingly.

Flow

The size of the wastewater collection system impacts no other effluent characteristic as much as it does the hourly variations in flow. Extreme hourly variations in single domestic service flows only gradually become attenuated with increasing numbers of connections. Small city wastewater flows fluctuate significantly more than those from larger cities. As shown in Figure 3, the 24-hour flow variations from a population of 5,000 could range from 0.4 to 2.5 times the average daily

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TABLE 5. TRACE ELEMENTS IN RAW MUNICIPAL WASTEWATER (mg/1)

Metal	Northeast[a]		Allegheny Co., PA[b]	Muncie, IN[c]
	Median	25-90% Range	Average	Average
Arsenic	--	--	--	--
Cadmium	<0.02	<0.02 - <.02	0.021	
Chromium	<0.05	<0.05 -0.30	0.095	0.24
Copper	0.10	0.05 -0.40	0.127	0.25
Iron	0.90	0.50 -2.50		
Lead	<0.20	<0.20 <0.20	0.119	0.93
Mercury	0.0013	0.0008-0.0045	--	--
Nickel	<0.10	<0.10 -0.20	0.078	0.14
Silver	<0.05	<0.05 -<0.05	--	--
Zinc	0.18	0.10 -0.76	0.648	1.15

a. 1970-72 sampling of New York, New Jersey, Connecticut by Interstate Sanitation Commission (3-3).

b. January-June 1973, Allegheny Co., Pennsylvania (3-4).

c. 1973 yearly average, Muncie, Indiana (3-4).

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TABLE 6. TRACE ELEMENTS IN MUNICIPAL EFFLUENTS

Element	Untreated wastewater[a], mg/l	Primary effluents, mg/l			Secondary effluents, mg/l		
		National[a]	Northern California[b]		National[a]	Northern California[c]	
			Average	Range		Average	Range
Arsenic	0.003	0.002	0.013	0.004-0.02	0.005-0.01	0.0007	0.0005-0.019
Cadmium	0.004-0.14	0.004-0.028	0.03	0.003-0.1	0.0002-<0.02	0.006	0.0005-0.01
Chromium	0.02-0.700	< 0.001-0.30	0.04	< 0.005-0.183	< 0.010-0.17	0.01	0.0005-0.03
Copper	0.02-3.36	0.024-0.13	0.085	0.04-0.125	0.05-0.22	0.056	0.01-0.12
Iron	0.9-3.54	0.41-0.83	--	--	0.04-3.89	--	--
Lead	0.05-1.27	0.016-0.11	0.1	0.02-0.2	0.0005-<0.20	0.038	0.01-0.06
Manganese	0.11-0.14	0.032-0.16	--	--	0.021-0.38	--	--
Mercury	0.002-0.044	0.009-0.035	0.0017	0.0002-0.004	0.0005-0.0015	0.0009	0-0.002
Nickel	0.002-0.105	0.063-0.20	0.15	0.01-0.4	< 0.10-0.149	0.06	0.03-0.2
Silver	--	--	0.02	0.005-0.05	--	0.19	0.002-0.034
Zinc	0.030-8.31	0.015-0.75	0.31	0.06-1.05	0.047-0.35	0.15	< 0.05-0.32

a. Source: reference 3-7.

b. Derived from 1978 end-of-year self-monitoring report averages for Benicia, Pacifica, Estero, Pittsburg, San Mateo, San Francisco Southeast, San Francisco Northeast and Sausalito-Marin City, California. Unpublished NPDES data.

c. Derived from 1978 end-or-year self-monitoring report averages for Hayward, Central Contra Costa, East Bay Municipal Utility, Novato-Ignacio, Petaluma, Sonoma Valley, Mountain View, North San Mateo County, Oro Loma and San Jose, California. Unpublished NPDES data.

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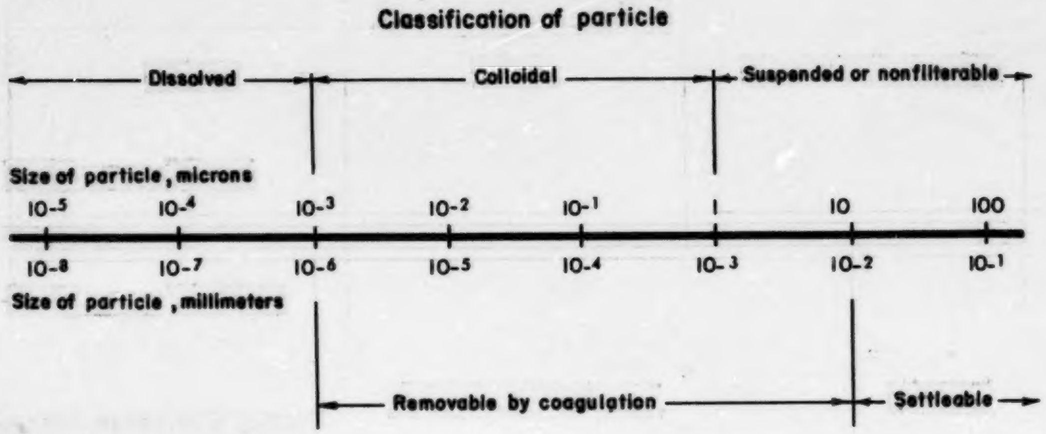
Classification of particle
Dissolved
Colloidal
Suspended or nonfilterable
Size of particle, microns
10-5
10-4
10-3
10-2
10-1
1
10
100
10-8
10-7
10-6
10-5
10-4
10-3
10-2
10-1
Size of particle, millimeters
Removable by coagulation
Settleable

Figure 2. Classification of particles found in water.[a]

a. Source: reference 3-1.

TABLE 7. SIZE DISTRIBUTION OF WASTEWATER PARTICLES[a]

Fraction	Size range, microns	Raw wastewater Total solids, mg/l	Raw wastewater Volatile matter, mg/l	Secondary effluent Total solids, mg/l	Secondary effluent Volatile matter, mg/l
Soluble	< 0.001	351	116	312	62
Colloidal	0.001-1	31	23	8	6
Supra-Colloidal	1-100	57	43	28	24
Settleable	> 100.	74	59	0	0

a. Source: reference 3-8.

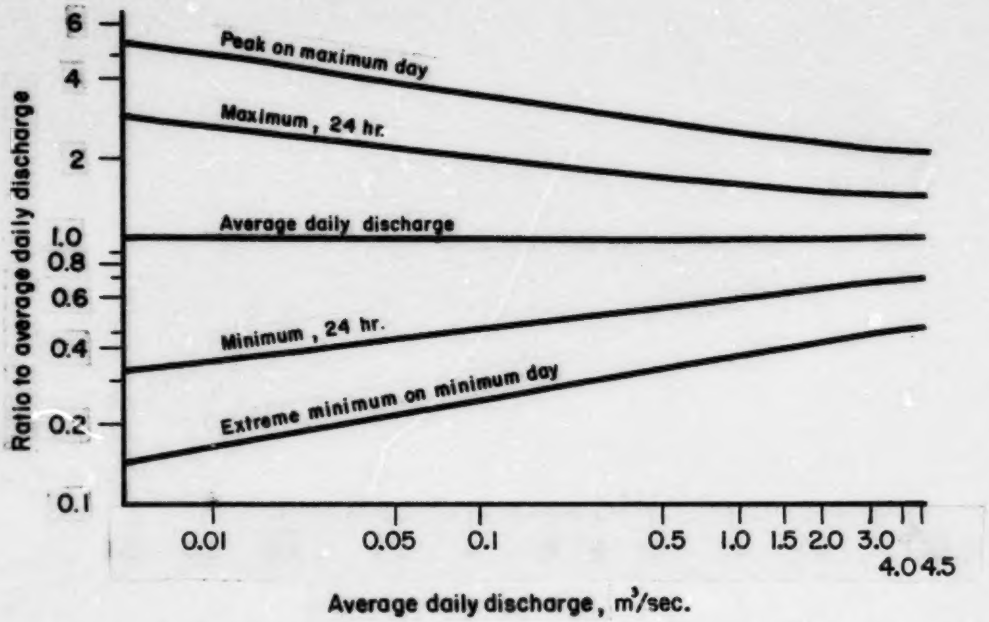

Figure 3. Variations in municipal wastewater flows.

Source: reference 3-1.

flow while for a population of one-half million, the range could be 0.7 to 1.5 times the average daily flow. Assuming minimum attenuation of the flow through the sewage treatment facility, these numbers represent typical variations in effluent loading upon a subsequent vegetative treatment system. However, as subsequent sections will point out, compared to stormwater runoff flows, treatment plant effluents are considerably less variable.

STORMWATER RUNOFF

Urban Runoff

Urban runoff is generated by rainfall upon land areas developed for residential, commercial or industrial uses and includes roadways. Urban runoff is generally considered to be surface runoff discharged by overland flow or through storm sewers. In a number of older cities, particularly in the eastern United States, sanitary sewers were designed to carry urban stormwater. The flow of these combined sewers, in excess of their carrying capacity or ultimate treatment capacity is combined sewer overflow (CSO).

Table 8 presents reported water quality from two studies on urban runoff and CSO's. Combined sewer overflows carry more organic wastes, nutrients and bacteria than urban runoff, due to the sanitary sewage component. However, urban runoff contains significantly higher percentages of inert suspended solids.

The quality of urban runoff water has been shown to be dependent upon the land uses in the contributing watershed. Table 9 provides a direct comparison of runoff qualities from industrial, commercial and residential neighborhoods in the Chicago metropolitan area. In general, commercial neighborhoods produce the poorest quality of urban runoff, but not by orders of magnitude. Also, light industrial neighborhoods generated runoff quality higher than for the other categories. This could be an unexpected finding unless it is kept in mind that light industrial areas, with landscaped grounds, warehouses and offices, more closely resemble low-density residential neighborhoods than heavy industrial areas with large factories, rail yards, shipments of raw materials and finished products, and possible direct pollutant emissions.

Table 10 presents urban runoff sampling results from the San Francisco Bay Area. In the eastern and central United States, where previously reported studies were conducted, storm runoff occurs during the summer and is quite different in character from gradual winter snowmelt. In the western states, summer is characterized by extended dry periods with most rainfall occurring during the period November to April. However, the results presented in Table 10, although collected during a period of unusally infrequent rainfall, are comparable to the Chicago data.

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TABLE 8. URBAN STORMWATER AND CSO QUALITY

	Pollutant concentrations, mg/l					
	Field[a]		Kaiser Engineers[b]			
			Urban stormwater[c]		CSO[d]	
Pollutant	Urban stormwater, range	CSO, range	mean[e]	std. dev.	mean[e]	std. dev.
BOD_5	11-500	100-500	27	25	108	36
COD	--	--	205	118	284	110
Suspended solids	500-11,300	100-1,500	608	616	372	275
Total solids	1,000-14,600	300-2,000	--	--	--	--
Organic nitrogen-N	0.9-16	3.5-30.1	--	--	--	--
Ammonium nitrogen-N	0.4-2.5	1.1-11.5	--	--	--	--
Total nitrogen-N	--	--	2.3	1.4	9	6
Phosphate-P	10-125	1-62	--	--	--	--
Total phosphorus-P	--	--	0.5	0.4	2.8	2.9
pH (units)	--	4.9-8.4	--	--	--	--
Total coliform (MPN/100 ml)	$2x10^3$-$14x10^5$	$5x10^4$-$30x10^6$	$3x10^5$	--	$6x10^6$	--
Fecal coliform (MPN/100 ml)	--	$5x10^4$-$11x10^6$	--	--	--	--
Chloride	200-25,000	--	--	--	--	--
Oils	10-110	--	--	--	--	--
Phenols	0-0.2	--	--	--	--	--
Lead	0-1.9	--	--	--	--	--

a. Source: reference 3-9.
b. Source: reference 3-10.
c. 20 cities, storm sewers and unsewered areas, unspecified locations.
d. 25 cities, combined sewer areas, unspecified locations.
e. Geometric mean.

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TABLE 9. COMPARISON OF URBAN RUNOFF BY LAND USE[a]

Constituent	Concentrations, mg/l									
	Single family		Multi-family		Commercial		Light Industrial		Highway	
	mean	range	mean	range	mean	range	mean	range	mean	range
BOD_5	17	2-80	16	6-41	22	7-53	14	6-38	14	2-29
COD	104	34-308	117	44-270	168	36-301	78	46-157	128	42-319
Total suspended solids	513	23-6,470	797	25-3,484	386	26-2,938	302	8-3,222	266	10-1,432
Volatile suspended solids	54	2-532	68	9-302	44	4-164	36	3-362	47	3-160
Kjeldahl nitrogen as N	1.7	0.4-8.0	2.1	1.3-7.4	2.0	0.8-4.5	1.4	0.5-3.6	1.2	0.1-3.7
Ammonium nitrogen as N	0.5	0.1-2.3	0.5	0.1-2.2	0.5	0.1-1.3	0.4	0.1-1.3	0.4	0.1-0.9
Nitrite and Nitrate as N	1.4	0.1-6.8	1.1	0.2-3.0	1.6	0.3-3.1	1.2	0.1-3.9	1.1	0.4-3.4
Soluble Phosphorus as P	0.14	0.01-0.65	0.23	0.04-0.69	0.24	0.02-2.18	0.19	0.01-1.48	0.07	0.02-0.13
pH (units)	--	6.4-8.4	--	6.5-8.5	--	6.8-8.5	--	6.6-8.7	--	6.5-8.2

a. Sampling of April-July 1977, Chicago Metropolitan Area (3-17).

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TABLE 10. SAN FRANCISCO BAY AREA URBAN RUNOFF MONITORING RESULTS GROUPED BY PREDOMINANT LAND USE[a]

Land use	% in that use	Number of samples	Average pollutant concentrations, mg/l BOD_5	SS	VSS	Total N	Total P	Population density, #/acre
Residential								
Alameda								
Castro Valley								
1971-1972	70(?)	36	6.7	904	--	2.0	0.7	--
1972-1973	--	59	31.0	945	--	3.7	0.8	--
1976-1977	70	33	25.7	339	--	--	--	21.4
Glen Echo	60	40	40.3	227	--	--	--	26.1
Peralta	--	26	24.9	--	--	2.7	0.7	--
Richmond	76	50	19.8	115	39	3.2	0.4	21.7
Contra Costa								
Pine Ck.	23	11	--	--	--	--	--	--
Rheem	75	4	21.0	211	28	4.5	0.6	--
San Ramon at Walnut Creek	19	3	8.5	129	19	1.4	0.4	--
Walnut Creek	34(?)	8	8.8	132	23	2.7	0.6	11.4
Napa								
York Street	90	22	22.0	81	36	1.2	0.4	14.0
San Mateo								
Brewster	100	1	7.0	145	54	3.5	0.02	14.0
Colma	21	1	16.0	207	43	3.5	0.3	--
Serramonte	81	1	14.0	85	28	1.3	0.2	--
Santa Clara								
Calabazas	38	46	19.4	557	88	4.5	0.9	18.2
Guadalupe	33	32	27.0	461	98	6.4	0.8	16.7
Commercial								
Santa Clara-San Jose	100	36	21.1	158	67	7.0	0.7	--
Industrial								
Santa Clara (light)	100	28	38.1	72	21	3.1	0.4	--
Santa Clara (heavy)	100	37	13.0	114	45	4.5	0.3	--
San Mateo-Bay front	80	1	16.0	90	26	1.3	0.2	--
Open								
Contra Costa-San Ramon Rudgear	94	3	6.8	220	38	3.8	0.6	--
Contra Costa-Little Pine	100	1	9.7	5200	550	2.6	2.4	--
Marin-Sleepy Hollow	80	41	4.3	87	16	2.4	0.2	--
Santa Clara-Berryessa	73	80	8.0	1134	146	3.6	1.0	--

a. Monitoring of 8 counties during 1976-77 rainy season, 1-7 storms per site (3-12).

Although there are differences in runoff quality based upon source, the differences are not extreme. It is therefore possible to assume a typical range of values for planning purposes although specific site characterization should be performed before actual design of any urban runoff treatment facilities. Unique local conditions may alter the mass emission of one or several runoff components. For example, the Association of Bay Area Governments found that in newly developing areas, construction-related erosion caused 15 to 20 times as much soil loss as background conditions, and in one watershed in Santa Clara County, 72 percent of the sediment leaving the basin originated from 6 percent of the land area [3-19]. Thus, the planner and designer must not only identify the unique contributing factors in a watershed but also should be aware that some factors, such as construction-related erosion, may be transitory and gone, or increasing, within several years.

Rural Runoff

The emphasis of this study is on urban runoff and wastewaters. However, the techniques of vegetative treatment are certainly applicable to rural or agricultural runoff. Table 11 presents typical values of agricultural runoff as collected from several sources. Forested land produces lower pollutant concentrations than urban runoff but as agricultural activities increase through the categories of rangeland, cropland and feedlots, the runoff quality significantly deteriorates in terms of suspended solids and nutrients. It would appear that remedial measures should be directed at solids control and reduction of nutrients.

Trace Elements in Urban Runoff

Data on trace elements in urban runoff are sparse. Table 12 presents concentrations of trace elements found in San Francisco Bay Area urban runoff. The highest concentrations found were of lead, zinc and arsenic and the lowest of silver, cadmium and mercury. However, these concentrations do not reflect the toxicity of the individual elements to specific organisms. Therefore, a program to remove toxic elements would depend upon the element and the target organisms to be protected or water quality standard to be achieved.

Table 13 contains the analysis of trace elements in stormwater solids filtered from samples collected from storm sewers. The most common element is iron at 28.4 g/kg followed by lead and zinc at 3.99 and 1.21 g/kg, respectively.

Hydrocarbons in Urban Runoff

Concern may arise about the hydrocarbon content of urban runoff. With potential sources ranging from oil spills and crankcase drippings to exhaust emissions and illegal dumping, urban runoff has the potential for direct, significant contamination of receiving waters by hydrocarbons.

TABLE 11. AGRICULTURAL RUNOFF CHARACTERISTICS

Land use	Data type	Concentrations, mg/l						
		BOD$_5$	Suspended solids	Volatile suspended solids	Total nitrogen-N	Nitrate nitrogen-N	Total phosphorus-P	Soluble phosphorus-P
Forested land[a]	range	--	--	--	0.3-1.8	0.1-1.3	0.01-0.11	--
Forested land[b]	avg.	2	34	7	0.5	0.2[d]	--	0.04
	range	1-5	20-51	3-13	0.2-0.6	0.1-0.4	--	0.01-0.06
Hilly, lightly grazed parkland[c]	avg.	5.1	1,290	133	1.6	--	2.4	--
	range	3.1-7.4	488-2,075	55-164	0.19-3.5	--	1.0-3.3	--
Rangeland , open and hilly[c]	avg.[b]	--	600	88	4.9	--	0.77	--
	range	--	228-1,010	26-138	1.3-8.1	--	0.27-1.9	--
wooded, hilly[c]	avg.	5.3	580	59	3.0	--	0.35	--
	range	5-5.9	222-924	46-77	2.3-3.4	--	<0.2-0.46	--
Agricultural cropland[a]		7	--	--	9	0.4	0.02-1.7	--
Agricultural cropland[b]	avg.	13	762	99	2.9	4.2	--	0.27
	range	3-36	5-6,100	1-883	0.1-12.9	0.3-23.5	--	0.02-1.54
Irrigation tile drainage, western U.S.[a]	range	--	--	--	2.1-19	1.9-19	0.01-0.3	--
Cropland tile drainage[a]	range	--	--	--	10-25	--	0.02-0.7	--
Seepage from stacked manure[a]	range	10,300-13,800	--	--	1,800-2,350	--	190-280	--
Feed lot runoff[a]	range	1,000-11,000	--	--	920-2,100	10-23	290-360	--

a. Source: reference 3-7.

b. Source: reference 3-17.

c. Source: reference 3-11.

d. Nitrite plus nitrate nitrogen.

TABLE 12. TRACE ELEMENTS IN URBAN RUNOFF[a]

	Flow weighted mean concentrations, mg/l								
Land use	Lead	Cadmium	Chromium	Silver	Nickel	Arsenic	Mercury	Copper	Zinc
Residential	0.61	0.01	0.02	0.005	0.09	0.21	0.01	0.15	0.58
Commercial	0.85	--	--	--	--	--	--	--	--
Industrial	1.3	--	--	--	--	--	--	--	--
Open and agricultural	0.15	0.007	0.007	--	--	--	--	--	--

a. 1976-77 sampling of San Francisco Bay Area counties (3-12).

TABLE 13. TRACE ELEMENTS IN STORMWATER SOLIDS[a]

Metal concentrations, mg/kg						
Lead	Chromium	Nickel	Copper	Zinc	Iron	Manganese
3,990	58	57	325	1,210	28,400	398

a. Stormwater solids <1,000 μm (3-13).

The two basic classes of hydrocarbons present in petroleum oils are the aliphatics and aromatics. Aliphatic hydrocarbons have the carbon atoms in an open-chain structure and are fatty in nature. Aromatic hydrocarbons have the carbon atoms arranged in a closed-ring structure, of which benzene is the elementary form.

Upon entering an aqueous environment, the chemical structure of the hydrocarbon determines its fate. Most hydrocarbons are insoluble and either adsorb to particles in the water or float on the water surface as a film. The low molecular weight hydrocarbons, both aliphatic and aromatic, are quite volatile and evaporate quickly. Often these compounds have evaporated almost completely before the runoff-causing storm event occurs. The hydrocarbons adsorbed to particulate matter settle to the bottom where microbial degradation can occur. The highly branched aliphatics and the remaining aromatic hydrocarbons biodegrade very slowly, if at all.

Table 14 presents the limited data available on hydrocarbons in urban runoff. They generally represent the low-volatility hydrocarbons that remain after some period of exposure to the air. Most references indicate that automobile crankcase drippings are responsible for the hydrocarbons in urban runoff [3-14, 3-15, 3-16].

The data in Table 14 show oil and grease concentrations (which include polar molecules of animal and vegetable origin and nonpolar molecules of petroleum and mineral origin) in urban runoff, ranging from 6 to 44 mg/l. A recent evaluation of 55 municipal wastewater treatment plants, 16 using trickling filters and 39 using some variation of activated sludge, showed a mean effluent oil and grease content (using varied extraction media) of 6 mg/l with a range of 0.1 to 61.8 mg/l, and 70 percent of all samples at less than 10 mg/l [3-20]. It, thus, appears that the oil and grease content of urban runoff is higher than that of wastewater treatment plant effluent.

Urban Stormwater Particle Size

Based upon studies in Michigan, urban stormwater particle sizes will be predominantly (89 percent by weight) below 100 microns [3-18]. This corresponds to the colloidal and supra-colloidal categories, where simple sedimentation is unlikely to produce significant removal of solids unless assisted by chemical or electrostatic attraction. Presumably, eroded soil from construction areas could substantially increase the settleable fraction of the urban runoff.

In Table 15, the urban runoff particle size distribution is compared to an analysis of street solids. The street solids analysis indicates that less than 25 percent of solids are in the nonsettleable or poorly settleable range. This apparent discrepancy can be reconciled by the explanation that street surface runoff represents only a portion of

TABLE 14. HYDROCARBONS IN URBAN RUNOFF

Source	Average suspended solids, mg/l	Hydrocarbon type	Average hydrocarbon concentrations, mg/l			
			Particulate	Soluble	Total	Range, mg/l
Philadelphia, Pa.[a]	87	Aliphatic	2.28	0.29	2.57	--
		Aromatic	1.01	0.11	1.12	--
		Total	3.29	0.40	3.69	--
San Francisco Bay Area[b]						
Residential	--	Total oil	--	--	6	5-8.3
Commercial	--	and grease	--	--	28	--
Industrial	--		--	--	25	5-33
Seattle, Wa.[c]						
Urban	--	Oil and grease[d]	--	--	10	1.0-16
		Aliphatic	--	--	1.2	0.2-7.5
		n-paraffin[e]	--	--	0.013	0.004-0.35
Highway bridge	--	Oil and grease[d]	--	--	27	0-96
		Aliphatic	--	--	12	6.0-24
		n-paraffin	--	--	0.10	0.025-0.25
Urban freeway	--	Oil and grease[d]	--	--	44	10-60
Philadelphia, Pa.[f]	--	Aromatic	1.25	0.06	--	--
		Total petroleum	4.28	0.25	--	--

a. 5 storms in 1974 (3-14).

b. Source: reference 3-11.

c. Source: reference 3-15.

d. Other Seattle studies referenced in 3-15.

e. n-paraffin = C_{13-36} except pristane C_{19} and phytane C_{20}

f. 3 storms in 1975 (3-16).

TABLE 15. SIZE DISTRIBUTION OF URBAN STORMWATER PARTICULATES[a]

Southeast Michigan 3 urban watersheds runoff average		EPA 5 cities street solids average	
size range (μm)	percent by weight	size range (μm)	percent by weight
<53	81.7	<43	13.5
53-106	7.0	43-104	11.0
107-250	4.1	105-246	17.6
251-850	4.3	247-840	22.2
>850	2.8	>840	35.6

a. Source: reference 3-18.

urban runoff pollutant sources, others being lawns, roofs, driveways, parking lots, inlets and catchbasins, parks and open areas. Thus, studies that characterize street sweepings do not accurately reflect true stormwater pollutant characteristics.

Urban Runoff Flow Characteristics

An important characteristic of urban runoff flow is its variability. While a municipal wastewater treatment facility may discharge an effluent that varies ±50 percent from average in a day, urban runoff will occur as intense runoff during short periods, often separated by very long dry periods. Table 16 lists the number of days per year that precipitation exceeds 0.25 mm (0.01 in) in several major cities. It is apparent that any vegetative treatment system for urban runoff must tolerate extended dry periods between rainfall events. When rainfall occurs during specific seasons, the vegetative treatment system must withstand drought (typically five months in the San Francisco Bay Area), must include storage of water, or have an alternate source of water to maintain vegetation.

In addition to accommodating significant variations in stormwater runoff frequency, a vegetative treatment system will experience variable flow during a storm event. Figure 4 shows typical hydrographs for developed and undeveloped watersheds. Storm runoff into a treatment system will rise to peak flow, then drop and taper off slowly during any single rainfall event. The effect of urban development, with its higher proportion of impervious surfaces, upon a watershed, is to decrease the time of concentration of flow and to increase total runoff from the watershed. This produces significantly higher discharge peaks and expands the range of hydraulic loading within which a treatment system must function.

Dry weather storm sewer discharges may be an unexpected source of water and pollutants. Although flows were not measured, recent dry weather sanitary surveys in several South San Francisco Bay cities uncovered significant storm sewer discharges [3-22]. Dry weather (no measurable rainfall for 14 preceding days) log mean total coliform levels of 2.1 to 6.5×10^4 MPN/100 ml approached wet weather levels of 2.5 to 12.6×10^4 MPN/100 ml for two major storm sewers. It is theorized that dry weather flow sources are ground water, domestic lawn watering and car washing, industrial dumping (particularly cooling waters), sanitary sewer cross-connections, and sanitary sewer pump station bypasses.

TABLE 16. ANNUAL DISTRIBUTION OF RAINFALL EVENTS[a]

	Number of days with 0.25 mm rainfall or more				
Month	Oakland	Denver	Boston	Miami	Chicago
January	11	6	12	6	10
February	10	6	11	5	10
March	9	8	12	6	12
April	6	9	12	6	13
May	3	10	12	10	12
June	1	9	11	15	10
July	0	9	9	16	9
August	1	8	10	16	8
September	1	6	9	17	8
October	4	5	9	15	7
November	7	5	11	8	10
December	10	5	10	6	10
Total	63	87	128	127	120

a. Source: reference 3-23.

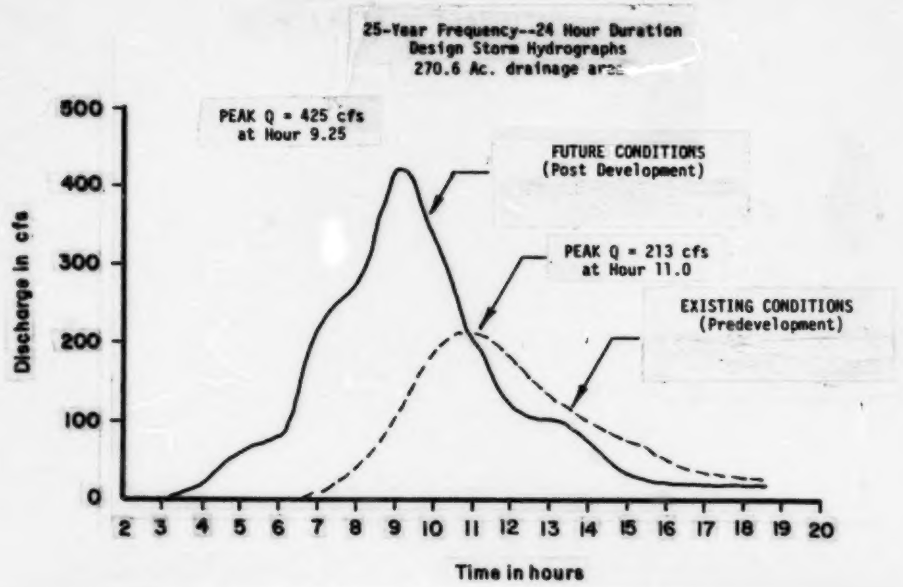

Figure 4. Typical small watershed hydrographs.[a]

a. Source: reference 3-21.

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SECTION 4

PHYSICAL AND CHEMICAL POLLUTANT REMOVAL MECHANISMS IN WETLANDS

A description of pollutant removal mechanisms that may operate in a marsh excludes few aquatic processes that can be found in any natural water body. All the mechanisms that are discussed in this section are not important in every marsh. Indeed, some are mutually exclusive, occur only seasonally, or are of secondary importance at most sites. However, most of these mechanisms are likely to occur to some degree in the majority of marshes.

This section describes pollutant removal processes that occur in natural waters and are likely to apply to marshes. Many have not been documented in marsh environments, largely due to the traditional neglect marsh research has received as compared with studies of rivers, lakes and oceans. Nevertheless, processes are discussed as long as they are likely to occur in some marshes to some degree. The types of marsh environments that would tend to favor specific processes are described in connection with each mechanism that does not have widespread occurrence.

POLLUTANT MASS BALANCE

Pollutants may enter a marsh via surface runoff or point source discharges. Other routes, e.g., through base flow, aerial fallout, and mobilized solid waste deposits, are not addressed here although the same treatment processes would be operative. For the purposes of this section, pollutants are defined as alien substances, and native substances in excess of their natural concentrations. Pollutants can be distinguished from the natural constituents of water by their source, since they arise from man's activities.

Surface runoff pollutants entering a marsh occur on the surface or in the water column, either dissolved, emulsified, or in particulate form. Likewise, surface water flowing from a marsh carries pollutants in the same ways. These substances are not necessarily the same as the inflow pollutants. Many types of transformations occur in marshes and the products may leave the marsh. Pollutants can be removed by a wetland system through three main routes: loss to the atmosphere, incorporation into sediments or biota, and degradation. Some products of pollutant degradation may be inert or nontoxic, while others continue to pose environmental hazards. For example, some readily oxidizable organic compounds are quickly converted to carbon dioxide and water. On the other hand, some pesticides are very stable or yield stable degradation products which are also toxic to the environment. Regardless of the nature of the products of pollutant transformations, they either flow out of the marsh, volatilize, or remain in the marsh, fixed in biota or sediments. A summary of the major pollutant removal mechanisms and the contaminants affected is presented in Table 17.

POLLUTANT LOSS TO ATMOSPHERE

Generally, the atmosphere acts as a sink rather than a source of pollutants to marshes. In some cases, however, aerial input may be an important factor, such as for lead in auto exhaust [4-3], solids from wind erosion of soil, and acid rain. Loss of pollutants to the atmosphere occurs by evaporation [4-12] and by aerosol formation [4-10]. Evaporation is by far the more significant of the two mechanisms for most marsh environments. The evaporation rate of a volatile substance is controlled by air and water temperature, wind speed, subsurface agitation, and the nature of any surface film that may be present. When the wind speed at the water surface is less than about 3 m/s, the volatilization rate is additively dependent on both wind speed and subsurface agitation. Above that value, wind speed dominates [4-6].

The presence of a surface film can decrease the rate of volatilization by acting as a barrier to volatile solutes in the water which are insoluble in the film. Alternatively, the surface film can scavenge substances from the bulk water and concentrate them at the surface where they, subsequently, are volatilized. Surface films occur in both freshwater and marine environments. They are composed of a highly variable array of compounds including fatty acids, esters, alcohols, lipids, hydrocarbons and proteinaceous materials. Surface films can contain dead fish, insects, and various living organisms and can be richer than the subsurface water in phosphorus, nitrogen, carbon, most heavy metals, chlorinated hydrocarbons and other materials [4-11, 4-12]. Concentrations of zinc, cadmium, lead and copper may reach 100 ppm in surface films, promoting reactions that may not otherwise occur and allowing ready entrance for the metals to the food web through surface feeding fish, insects and other organisms.

Environmentally important pollutants for which volatilization may be a significant fate include oils [4-19, 4-28, 4-48], chlorinated hydrocarbons [4-10], 2,4-D esters [4-62], and elemental mercury [4-47]. The multitude of compounds making up petroleum oils vary greatly in their volatility. The acutely toxic, low-boiling-point alkanes evaporate quickly while the atmospheric loss of most large molecular weight petroleum hydrocarbons is negligible.

Aerosol formation can be important in marshes only when a strong wind blows over a long inundated fetch. The wind generates a foam from the surface film which is enriched in pollutants [4-11]. As the foam bubbles burst, droplets are ejected into the air. Water and other volatile substances quickly evaporate from the droplets, leaving the aerosol residue.

TABLE 17. PHYSICAL AND CHEMICAL POLLUTANT REMOVAL MECHANISMS IN WETLAND AND AQUATIC SYSTEMS

	Pollutant affected									
Mechanism	Settleable solids	Colloidal solids	Organic compounds	Petroleum hydrocarbons	Nitrogen	Phosphorus	Heavy metals	Bacteria and viruses	Halogenated hydrocarbons	Description
Physical										
Evaporation			X	X			X[a]		X	Volatilization and aerosol formation
Sedimentation	X	X		X	X		X[b]		X	Gravitational settling of particles and adsorbed pollutants
Emulsification		X	X	X			X[a]		X	Suspension of chemicals that are sparingly soluble in water within an aqueous environment
Adsorption		X	X	X			X	X	X	Electrostatic attraction, Van der Waals force
Filtration	X	X						X		Mechanical filtration of particles through substrate, roots or animal systems
Chemical										
Chelation						X	X			Formation of metal complexes through ligands
Precipitation				X		X	X			Formation of or coprecipitation of insoluble compounds
Decomposition			X	X	X	X	X	X	X	Alteration of less stable compounds by oxidation, reduction, hydrolysis or photochemical reaction
Chemical adsorption			X	X	X	X	X		X	Covalent bonding, hydrogen bond formation, hydrophobic interaction

a. Significant only for mercury.
b. Not significant for manganese and mercury.

SEDIMENTATION

Sedimentation is one of the most important mechanisms by which particulate pollutants are removed from the water column. Particulate matter may be considered a pollutant, either because it consists wholly of pollutant material or because pollutants are adsorbed to the surface of naturally occurring particles. The mechanisms by which substances adhere to the surface of solids are dealt with later in this section. The following describes the mechanics of particulate settling.

Sedimentation Theory

Three physical processes cause the movement of particles in water: (a) Brownian motion, (b) fluid shear, and (c) gravity [4-40]. Brownian motion, also called molecular diffusion or thermal agitation, is important for extremely small particles with diameters less than about 1μ [4-52]. Solids in this size range exhibit random vibratory motion when suspended in water. This movement has little effect on the removal of solids from the water column since there is no net movement toward the bottom. However, in the microzone at the sediment/water interface this motion may bring solids close enough to larger particulates for other forces to effect adsorption.

Gravity moves particles with a specific gravity greater than one downward while turbulence tends to cause resuspension. The competition between these two forces determines the rate of settling for any size/density class of suspended matter. Suspended solids in natural waters and wastewaters range in diameter from 0.005μ to 100μ , but for particles having diameters less than about 10μ , the terminal settling velocity is less than 0.01 cm/s [4-52]. Generally, the larger the particle, the higher the settling velocity.

In natural waters, particulate matter usually occurs as a mineral in an organic matrix [4-4, 4-5, 4-8, 4-24]. For example, the inorganic portion may be clays, quartz particles or hydrous metal oxides while the organic part is usually humic or fulvic acids. The organic/mineral aggregates have a loosely articulated, globular structure, often with void spaces [4-8]. The clay particles in fresh waters usually have negative charges. Ions in the water surround negatively-charged particles, forming an electrically neutral "double layer." Repulsion between like charges prevents the particles from agglomerating into larger, more readily settling solids. The degree and rate of agglomeration of particles depends upon the frequency of collisions between particles in the water and the efficiency of the collisions. Collision efficiency is a term for the likelihood that a collision will result in the particles involved adhering to each other. Solutes in the water dramatically affect collision efficiency. Solutes that adsorb to the surface of particles can increase, decrease or reverse the surface charge. In addition, dissolved ions that do not interact with the colloidal particle can increase collision efficiency by compressing the diffuse part of the electric double layer with counter ions, allowing Van der Waals forces to cause agglomeration. An example of this is the enhanced settling of suspended matter in estuaries [4-8]. In fresh water the collision efficiency is only 0.0001 to 0.000001, but in sea water it rises to 0.1 to 1.0. Thus, sediment-laden freshwater flows entering an estuarine system should undergo enhanced solids deposition as compared to an otherwise completely freshwater system [4-63].

As mentioned above, collision frequency is also important in the sedimentation process. Quiescent bodies of water, such as marshes and other wetlands, often provide long detention times. Collision frequencies caused by Brownian motion for very small particles are very low, but with long detention times there are sufficient contact opportunities for significant agglomeration and settling of finely divided solids.

The nature of flow patterns through wetlands has been found to influence deposition of suspended matter and overall pollutant removal effectiveness. Sheet flow and meandering channels lead to large effective flow areas and best deposition rates [4-64, 4-65, 4-66]. Studies with artificial wetlands at the Brookhaven National Laboratory point to the importance of structural diversity, i.e., combining marshes, meadows, ponds, etc. to promote circuitous flow and greater deposition of particulate matter [4-65].

Wetlands generally act as detention and regulating areas for streamflows, with beneficial effects on downstream soil erosion. Within the wetland itself, however, high flow rates associated with storm episodes, seasonal runoff or regulated discharges may resuspend previously deposited and decomposing organic matter and wash it out of the wetland system [4-67, 4-68, 4-69].

Sedimentation of Pollutants

Sedimentation of pollutant particulates or pollutants adhering to the surface of particles can be the primary mechanism for removal of these substances from the water column. This process has been reported as important for removal of particulate nitrogen [4-33], oils [4-17], chlorinated hydrocarbons [4-5, 4-36], and metals, except for manganese and nickel [4-8, 4-43]. The chemical character of marsh water and the nature of the suspended solids determine the extent of pollutant adsorption onto the surface of the solids. Since the water chemistry can vary widely among marshes, an important mechanism at one location may be secondary at another. Factors affecting adsorption of various substances to suspended solids and bottom sediments are addressed later in this section.

EMULSIFICATION

Emulsions are finely dispersed colloidal mixtures of two or more immiscible liquids. Emulsions are inherently unstable and the liquids tend to coalesce unless an emulsifying agent is present to lower the interfacial energy between the immiscible liquids. In natural systems, the most common occurrence is the presence of minute droplets of oil in water [4-19, 4-48]. The stabilizing agent is a soap, detergent, or naturally occurring organic molecule, one end of which is polar and the other nonpolar.

Emulsions are formed when highly turbulent conditions exist in the presence of a suitable interfacial agent. This situation may arise at waterfalls, cataracts, hydraulic jumps in a wastewater collection system, or through the action of breaking waves. If stabilized, the emulsion may persist into downstream water bodies such as marshes, lakes, rivers or estuaries.

There are three principal mechanisms that destabilize emulsions in natural waters [4-55]:

(1) A change in the chemical character of the water, i.e., temperature, pH, ionic strength or presence of certain solutes, may reduce the ability of the agent to stabilize the emulsion. Salinity increases, as would occur when freshwater enters a salt marsh, aid the coalescence of the emulsion colloids into larger drops.

(2) An increase in the concentration of oil decreases the emulsion stability. This could occur when a marsh volume is reduced by evaporation, thereby lowering the water-to-oil ratio.

(3) Freezing and thawing of an emulsion usually causes it to break-up [4-55].

Emulsion colloids are important to the fate of pollutants in wetlands because, aside from containing toxic oils, they tend to accumulate environmentally significant chemicals that are sparsely soluble in water. This is important for pollutants such as mercury, polychlorinated biphenyls (PCB's), and some pesticides [4-47].

ADSORPTION

The principal means of removal of dissolved pollutants by adsorption in natural waters is adhesion to suspended solids or bottom sediments. The solutes are bound to the solid phase by three main mechanisms [4-52, 4-37]:

(1) Electrostatic attraction or repulsion;
(2) Physical reaction (Van der Waals forces, hydrogen bonding); and
(3) Chemical reaction.

Extensive contact between wastewater flows and wetland soils allows these interactions to occur. The most effective results are attained in seepage wetlands where a large percentage of the flow passes through the soil complex prior to entering ground waters or discharging from the wetland [4-68]. In wetlands having poor soil permeability, such as peatlands, shallow water depths and long residence times promote greater contact with surface soils and increased pollutant removal efficiencies [4-64].

Hickok and others have found the need to promote maximum interaction of runoff and wastewaters with mineral soils for attainment of high levels of phosphorus removal in wetlands [4-70, 4-71]. In addition to low water depths, even distribution of water, such as sheet flow, enhances soil-water contact over larger effective areas of the wetland [4-72].

Seepage wetlands function similarly to overland flow systems, with nearly complete phosphorus removal through soil processes [4-68]. In such systems, unusually high runoff flows may pose problems by causing overflows and bypassing of seepage routes [4-68]. Similarly, in flow-through wetlands, high runoff events reduce opportunity for soil contact and, as noted earlier, may be detrimental by washing sediments, organic matter and other constituents from the confines of the wetland.

A great number of substances adsorb to solids under conditions that may be found in wetlands. This phenomenon is well-documented in the literature:

- o organic compounds [4-4, 4-13];
- o petroleum and other hydrocarbons [4-17, 4-19, 4-21, 4-37, 4-48, 4-57];
- o halogenated hydrocarbons [4-1, 4-10, 4-25, 4-27, 4-47, 4-62];
- o ammonium [4-44, 4-45];
- o phosphorus [4-14, 4-20, 4-30, 4-51];
- o heavy metals [4-3, 4-8, 4-9, 4-15, 4-16, 4-24, 4-32, 4-34, 4-39, 4-43, 4-46, 4-47, 4-50];
- o bacteria and viruses [4-18, 4-47].

Petroleum and Other Organic Compounds

The importance of sedimentation as a mechanism for removing adsorbed petroleum from the water column was discussed earlier. Emulsified oils can adsorb to particles but physical agglomeration inhibits adsorption [4-17]. Structural properties of organic compounds affect the capacity for adsorption. The following general trends describe the tendency of organic compounds, having different chemical structures, to adsorb onto particulates in an aqueous environment [4-37]:

(1) Increasing with molecular weight as a homologous series is ascended, unless the molecule is so large as to be filtered out by small carbon pores.

(2) Decreasing with the compound's polarity, and hence, solubility;

(3) Decreasing with the position of substitution of hydroxy and amino groups on benzoic acids in the order: ortho, para, and meta;

In addition, the character of the particle/oil/water system affects the degree of sorption as follows [4-17]:

(1) Adsorption is greater with decreasing size of the sorbent particles, up to the point where the small pore size causes interference;

(2) Adsorption increases with increasing content of organic matter in the particles;

(3) With very low organic compound concentrations in the water, little adsorption occurs on inorganic particles such as clays.

Most halogenated hydrocarbons and organic pesticides adsorb strongly to particulate matter. In the case of the pesticide 2,4-D esters, adsorption inhibits evaporative loss from the water which might otherwise occur [4-62].

Ammonium

In salt water clay sediments, ammonium adsorption occurs predominantly onto organic matter [4-44]. This adsorption is rapid and reversible. When adsorption of ammonium to clays does occur, two mechanisms may apply: (a) ion exchange, or (b) attachment to the interlayers of the clay structure where replacement by other cations does not readily take place.

Phosphorus

Phosphate in freshwater sediments is closely associated with iron and aluminum by adsorption onto hydrous oxides of these two elements [4-51]. This adsorptive capacity is sensitive to both pH and redox potential [4-30]. Binding of phosphate decreases with increasing pH and decreasing redox potential. These two factors can strongly influence the migration of phosphate between sediments and the overlying water [4-14, 4-30, 4-51]. Phosphate and iron are released in deeper sediments when biological processes produce anoxic conditions. Dissolved phosphate and ferrous iron diffuse toward the sediment/water interface where the iron is oxidized and forms ferric complexes in the aerobic microzone. These complexes adsorb the phosphate, trapping it in an enriched layer at the top of the sediment. If the sediments become completely anoxic, the ferric complexes break down, releasing the phosphate.

Heavy Metals

Many heavy metals readily adsorb to sediment and water column particulates in wetland environments [4-24]. Ionic strength, pH, temperature, redox potential and the presence of competing cations can each affect the degree of surface bonding. An exposition on the varied conditions promoting metal adsorption is beyond the scope of this section. The reader is referred to the authoritative book by Stumm and Morgan [4-53] for a thorough treatment of the subject. To illustrate the types of processes defining the fate of heavy metals in wetlands, the following examples are presented:

(1) Cadmium is better adsorbed to suspended solids in low ionic strength water than in high ionic strength water under aerobic conditions [4-43]. In anoxic conditions, cadmium exchanges with iron at the surface of ferrous sulfide substrates [4-15].

(2) Copper, unlike cadmium, is adsorbed more strongly in high ionic strength water than in low ionic strength water [4-43].

(3) Lead adsorbs to metal-organic complexes and to clay minerals [4-46].

(4) Mercury readily adsorbs to solids but the route is frequently from oil rather than from the water [4-47].

(5) Zinc adsorbs to colloidal hydrous oxides in salt water environments [4-16].

Bacteria and Viruses

Because of adhesive organic compounds or electrostatic charges on their surfaces, microorganisms attach to particles in the water. The adsorption of viruses to suspended soil particles is a function of pH and dissolved inorganic and organic materials [4-18]. Viruses can remain for many months in the soil matrix and adsorption may even prolong their survival [4-18]. Desorption may occur in response to changing conditions, such as the presence of competing organic matter for the adsorption sites.

CHELATION

Chelation is the process by which metals are strongly bound in a complex with anionic molecules, either inorganic or organic. The anionic part of the complex is called the ligand. There are many sources of ligands in wetland environments. Salt waters are rich in important inorganic ligands, including chloride, sulfate and bicarbonate, which can form soluble complexes with trace metals. This chelation mechanism competes with adsorption of the metals onto the surfaces of iron and manganese hydrous oxides [4-43].

Organic ligands are also very common in natural waters. Partial decomposition of organic matter produces a variety of low molecular weight acids which chelate metals, including amino, butyric, citric, formic, 2-keto-gluconic, malic, oxalic, tartaric, and a variety of lichen acids [4-24]. In addition to forming complexes, these acids lower the pH. This promotes the dissolution of metals from solids, making them available for chelation. Large molecular weight organic compounds that are found naturally also are important ligands. Primarily, these are humic and fulvic acids [4-24, 4-46].

Many microorganisms actively excrete organic substances to chelate trace metals [4-26, 4-34, 4-35, 4-54]. These are proteinaceous compounds, such as amino acids, which have electron-rich functional groups which bind metal cations [4-56]. The ligands bind essential trace metals so that they are available for uptake by the organism. Iron binding is a good example of this process. In aerobic waters iron is in the ferric state and quite insoluble, or in the ferrous state and slowly oxidizing to ferric [4-73]. By complexing ferric colloids, microbial exudates keep the iron in the water column where it can be used as a nutrient. Although iron is complexed more strongly, these same exudates readily chelate copper. Copper is very toxic to microorganisms as a free ion but when complexed, the toxicity disappears.

There are ligands in most wastewaters and stormwaters. The sources of these ligands can be industrial, domestic, and agricultural chemicals. A good example of this is iron fertilizer. Ethylenediaminetetraacetic acid (EDTA) is present in fertilizer to permit uptake of the nutrient by plants [4-31]. Through runoff transport, EDTA and other ligands can affect distribution of metals in wetlands and uptake by vegetation.

Chelation is important to the mobility of aluminum, cadmium, calcium, chromium, copper, iron, lead, manganese, mercury, nickel and zinc. In addition to dissolving inorganic phosphate compounds, a number of organic acids may act as chelating agents, complexing calcium, iron, manganese, and aluminum. Such reactions can result in additional solubilization of phosphorus [4-14].

PRECIPITATION AND DISSOLUTION

Metals dissolve or precipitate in response to the changing aquatic environment. In anaerobic waters, insoluble ferric iron is converted to ferrous iron, which readily dissolves. In the presence of sulfide, which is common in anaerobic sediments, ferrous iron is precipitated as iron sulfide [4-29, 4-41]. Other metals, including cadmium, copper, lead, mercury, silver and zinc, form insoluble sulfides under reducing conditions common in wetlands [4-15, 4-16, 4-38]. In aerobic environments, aluminum, cadmium, chromium, iron, manganese and zinc form oxides and hydroxides [4-15, 4-16, 4-38, 4-51]. Dissolution of minerals that are more soluble at low pH often occurs in organic-rich sediments when acids are released by partial degradation of organic matter [4-24]. phosphate release in the aquatic environment This process is a common cause of phosphate release in the aquatic environment [4-7, 4-14].

OXIDATION AND REDUCTION

Oxidation and reduction always occur simultaneously. Oxidation is the loss of electrons and reduction is the gain of electrons. Metals are common catalysts for oxidation-reduction reactions, but more frequently enzymes associated with microorganisms catalyze transformations of this type. For some organic compounds oxidation reduction reactions are initiated by exposure to light (in wetland environments, sunlight). In some cases, water must be included to effect a balance, but this occurs naturally in aqueous solutions. Each of these reactions is described below.

Oxidation

Oxidation can occur in either the presence or absence of oxygen. Autooxidation is the reaction of oxidizable material with molecular oxygen. An example important to wetlands is the oxidation of hydrogen sulfide to sulfates. This reaction is catalyzed in both fresh and salt waters by metals such as cobalt, copper, iron, manganese and nickel [4-22]. Oxidation of organic matter is constantly taking place in all wetland systems. The most important route is by biological mediation. When oxygen is present, the degradation can proceed completely so that carbon dioxide and water are the products. Some naturally occurring organic matter resist complete oxidation and refractory substances remain. An important mechanism for the release of metals to water is the oxidation of the organic portion of particulate matter having associated trace metals.

In wetland sediments having high organic matter content, molecular oxygen is usually not present. Microbes can use nitrate and sulfate oxygen in these circumstances to oxidize organic matter. In general, readily biodegradable organic matter is anaerobically oxidized through a series of microbial transformations so that methane and carbon dioxide are the final products [4-23].

Other compounds are commonly oxidized in wetlands. Organic nitrogen is sequentially oxidized by bacteria in aerobic environments to ammonia, nitrite and nitrate [4-2]. Iron sulfide and pyrite are oxidized biologically to elemental sulfur, thiosulfate, polythiosulfate, sulfite, and sulfate [4-23, 4-41]. Arsenite is oxidized by microorganisms to arsenate [4-42]. Soluble ferrous ions, which diffuse into aerobic zones from underlying sediments, are oxidized to ferric oxides and hydroxides [4-30, 4-43] and chromium (III), which is common in rocks, is slowly oxidized to chromium (VI) [4-8]. Other metals are similarly oxidized [4-43].

Reduction

Chemical reduction of organic matter primarily occurs within plants and photosynthetic bacteria and is made possible by solar energy captured in photosynthesis. Chemoautotrophic bacteria are an exception to this reduction process and are of lesser significance. These processes are not discussed here as they are not pollutant removal mechanisms.

Several compounds are reduced in anoxic environments to provide oxygen for the oxidation of organic matter. Usually either nitrate or sulfate is the oxygen-rich agent that is reduced. Nitrate is reduced to nitrite, then to gaseous elemental nitrogen; sulfate is reduced to hydrogen sulfide, iron sulfide, and pyrite. Under anoxic conditions, some metals are reduced to lower oxidation states.

Hydrolysis

One of the most important hydrolytic reactions in natural waters is the conversion between ammonia and ammonium. When the water molecule is split, the hydrogen ion unites with ammonia while the hydroxyl radical remains in solution. This tends to raise the pH and provide a more favorable environment for biological oxidation of nitrogen [4-2].

Hydrolysis of some organic compounds occurs in wetlands. This reaction is particularly significant for materials that may be toxic to microorganisms. Hydrolysis may be a mechanism for removal of chlorinated alkanes from waters although the process may take months [4-10]. Metals are sometimes necessary to catalyze hydrolysis. For example, cobalt and copper can catalyze the hydrolysis of glycine methyl ester [4-22]. A few other illustrations of the hydrolysis of compounds having environmental significance are:

(1) The acid-based catalyzed reactions of phthalate esters and phosphate esters to yield organic acids and alcohols, which are biodegradable [4-13];

(2) Hydrolysis of the insecticides methoxychlor and dichlorodiphenyltrichloroethane (DDT) to dichlorodiphenylethylene (DDE) [4-60];

(3) Hydrolysis of 2,4-D esters to phenols [4-62].

Photochemical Reaction

The two principal types of photochemical reactions that occur in natural water bodies are [4-13, 4-61]:

(1) Direct photolysis, which is the absorption of light by a compound, followed by its disintegration to more reactive fragments. When one of the free radicals created is the singlet oxygen, which is a more reactive oxidizer than molecular oxygen, the process is called photochemical oxidation.

(2) Photosensitized oxidation, which is also called indirect or sensitized photolysis. This occurs when the singlet oxygen formed by photolysis initiates the oxidation of a substance other than the originally excited compound.

The extent of photochemical reactions is a function of the solar spectral irradiance at the water surface, the transfer of the light from air to water, and transmission through the water. In wetland waters, attenuation of light by dissolved natural organics can restrict these reactions to the upper layers. Scattering of light by suspended matter is usually much less important than absorbance [4-61]. Because the intensity of light usually drops with depth precipitously in wetland waters, the surface film is where most photochemical reactions take place [4-28, 4-62].

A number of examples illustrate photochemical reactions that can participate in the removal of compounds of environmental concern:

(1) EDTA is rapidly photodegraded in both acidic and basic waters [4-31];

(2) Ultraviolet irradiation of hydrocarbons found in No. 2 fuel oil produces relatively soluble oxygenated compounds, including reactive peroxides, phenols and carbonyl compounds [4-28];

(3) Photodegradation can shorten the life of unsaturated chlorinated alkanes but probably does not occur for their saturated counterparts [4-10];

(4) The polynuclear aromatic hydrocarbon, benzanthracene, is photodegraded in salt waters [4-21];

(5) Although usually slow, many pesticides undergo photodegradation including 2,4-D esters [4-61, 4-25], malathion [4-59], methoxychlor and DDT [4-61].

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SECTION 5

BIOCHEMICAL POLLUTANT REMOVAL MECHANISMS IN VEGETATIVE SYSTEMS

Vegetative systems have attracted attention as natural sinks for contaminants and as potential components for wastewater treatment systems [5-1 through 5-10]. Wetland systems are characterized by:

- o high plant productivity and nutrient needs;
- o high decomposition activity;
- o large adsorptive areas in the substrates;
- o low oxygen content of the sediments.

Dryland or upland systems are characterized by:

- o high plant productivity and nutrient requirements under irrigation;
- o large adsorptive areas in the substrates;
- o high physical, chemical and microbial activity in the soil mantle;
- o periodic reaeration of the soil layers.

These properties seem to provide vegetative systems with the ability to degrade and eliminate contaminants in wastewaters.

Changes in water quality in a variety of natural habitats, such as freshwater marshes, bogs, swamps, brackish and saltwater marshes, as well as artificially constructed wetland systems and managed grassland systems, have been studied. Although each habitat differs, these systems have been generally found to remove nitrogenous and phosphorus nutrients from wastewaters. Suspended and particulate matter are filtered out by plant roots or settle on the plant and soil surfaces. Petroleum hydrocarbons accumulate on surface soils and may be actively broken down. Chlorinated hydrocarbons show strong attractions to sediments and may be degraded by microorganism activity. Other contaminants that may be reduced are coliform bacteria and wastes that deplete oxygen from the system (as measured by BOD). Although little is known about the upper limits of wetland and dryland waste-processing abilities, studies of systems receiving pollutants continuously show that vegetative systems can function well even in severely polluted areas.

Vegetative systems possess a variety of ecological features that adapt them to gradual environmental changes. These features can also allow the vegetative systems to survive the added stresses of processing waste effluents up to some limits. Under certain conditions, wastewater input, particularly the nutrient components, can enhance the productivity of a system.

Since the majority of waterborne contaminants are adsorbed onto particulate matter, sedimentation of particulates effectively scrubs the water column, and prevents dispersal of the pollutants. These organic-rich sediments are further bound in place by plant roots. The tendency for these sediment layers to become anaerobic is probably the major factor involved in the retention of various chemical species [5-10]. Reducing environments allow the conversion of heavy metals into relatively insoluble sulfides and promote the removal of nitrate nitrogen through denitrification.

Vegetative systems possess a variety of mechanisms for obtaining nutrients and other elements from their environments under changing conditions. Through interaction with the various soil layers, water and air interfaces, plants can increase the overall capacity of a system to retain or remove pollutants. Since the primary mechanisms for pollutant removal in a wetland system are physical and chemical interactions that cause the contaminants to settle or be drawn out of the water column into the sediments, plant uptake of pollutants, particularly from the sediments, frees more exchange sites for further pollutant interaction and accumulation. Some researchers theorize that pollutant removal in aquatic systems is due, primarily, to bacterial metabolism and physical sedimentation (5-224). However, even they concede that plants provide surfaces for bacterial growth, filtration and adsorption of solids, attenuation of sunlight and nutrient absorption to retard growth of algae, and uptake of heavy metals. The biochemical processes related to wastewater renovation that will be discussed in this section are:

- pollutant uptake processes;
- plant adaptation mechanisms;
- nutrient cycling;
- heavy metal removal potentials;
- tolerances to other pollutants.

POLLUTANT UPTAKE PROCESSES

Pollutants in ionic form can be actively taken up by plants and accumulated in concentrations in excess of their environment. Plant cell membranes are not permeable to free ions of elements. Ions can only be transported across the cell membrane into the plasma through carriers which are active subunits of the membrane. Carriers have been hypothesized to be enzymes in that they have active sites that are specific for particular types of ions. Various carrier-transport mechanisms appear to function at different temperature levels [5-11]. Environmental conditions, such as increased light, temperature and carbohydrate energy sources, generally promote ion transport, whereas anaerobic conditions may inhibit absorption of specific ions. The physical and chemical mechanisms occurring within an aquatic environment that immobilize, liberate or facilitate uptake of pollutants were discussed in Section 4.

The pollutant uptake processes to be discussed below include:

- uptake through the plant-soil interface;
- uptake through the plant-water interface;
- translocation through plant systems;
- differential uptake vs nonspecific uptake;
- immobilization in the litter zone.

Uptake through Plant-Soil Interface

Vascular plants actively absorb ions from the soil through the root systems. Rooted aquatic plants can also absorb ions through roots, rhizomes and holdfasts. Buried portions of shoots and leaves are also capable of uptake, particularly of nutrients [5-12]. Uptake capability is usually directly proportional to the volume of belowground roots except in some submerged and floating plant species which have poor root systems.

Upland vegetation requires an aerated soil zone around the roots for proper plant growth. The rhizosphere or immediate zone around the roots shelters aerobic, nitrifying bacteria which convert ammonia-nitrogen to nitrate-nitrogen, the form most readily taken up by plants. Wetland vegetation, on the other hand, typically grows in saturated anaerobic substrates. These conditions favor ammonification and denitrification which result in high ammonia-nitrogen and low nitrate-nitrogen levels in the sediments and water. Wetland vegetation is typically adapted to use not only nitrate but also ammonia and ammonium as nitrogen sources.

In a wetland or aquatic environment, pollutants such as heavy metal ions can adsorb onto particulates or form complexes with inorganic phosphorus and settle down to the sediment layers. Concentrations of these pollutants are highest in the top layers of sediments and, unless they are immobilized into nonsoluble forms, provide a rich uptake source for vascular aquatic plants. The various processes immobilizing pollutants will be discussed later in this section.

Uptake through Plant-Water Interface

Most species of wetland vegetation are able to absorb some nutrients and ionic compounds from the water via shoots and leaves, although the primary uptake would still be through the roots at the plant-soil interface. Submerged plant species, particularly those which have weak or poorly-developed root systems, are capable of absorbing the bulk of their nitrogen requirements directly through the leaves [5-13, 5-14, 5-15]. Uptake of nutrients and other compounds is probably affected by the same factors that influence other plant physiologic processes, such as light, temperature and dissolved oxygen.

Translocation through Plant Vascular System

Movement of nutrients and other compounds from the root uptake sites through a plant vascular system can increase the capacity of a vegetative system to adsorb pollutants. The most rapid uptake usually occurs during the growing season when nutrients are translocated from the roots to the shoots and leaves. Peak uptake generally coincides with the period of peak biomass production [5-16, 5-17, 5-18].

With the death of emergent and aquatic vegetation in the fall (senescence), translocation from aboveground to belowground parts and winter storage of some nutrients and other compounds occur in wetland vegetation [5-16, 5-19, 5-20]. At the start of the next growing season, depending on the species, nutrient levels may have decreased in the roots and rhizomes and become concentrated in the shoots, probably preparatory for the period of new growth. Thus, at the start of the growing season, the aboveground parts (the shoots) will contain the highest nutrient concentrations [5-16, 5-19, 5-21]. Whether these high concentrations are sustained solely from the belowground plant resources or are a result of immediate uptake is not certain. Data indicate that root uptake could be sufficient to account for these concentrations [5-19]. It should be noted that peak accumulation in belowground parts generally does not occur until aboveground parts have reached their peak concentrations [5-16].

In some fast-growing plant species such as grasses, uptake of pollutants such as heavy metals increases with increasing heavy metal concentrations in the soil and water [5-22, 5-23]. In reed canarygrass, lead is translocated from the roots to the aboveground parts. However, unlike other emergent macrophytes, translocation back to the belowground parts does not occur when the plant senesces in the fall. Instead, as the aboveground mass falls to the ground and decomposes, the accumulated pollutants and other compounds are gradually released back to the soil and water.

Differential Uptake vs. Nonspecific Uptake

Certain plants accumulate dissolved materials, including trace contaminants, that are not required for plant growth or function. This nonspecificity is a function of general nutrient uptake processes where some minerals, such as strontium and calcium, are interchangeable ions in plant metabolism. In other cases, the accumulation of heavy metals without apparent toxic effect may be due to the presence of chelating compounds which combine with the metal ions to form harmless complexes. An example is Potamogeton pectinatus, a lead-tolerant plant which takes up lead readily. Incorporation into the cell wall renders the lead inactive and harmless to the plant [5-24].

Trace contaminants can be taken up and preferentially stored in different plant parts. In pondweed (Potamogeton spp.), most heavy metals tend to concentrate in the roots and rhizomes, while zinc accumulates in the stems and leaves. Tolerance of a heavy metal varies for any single plant species and tolerance of one metal does not necessarily indicate tolerance for others. In general, emergent aquatic plants have lower heavy metal content than floating or submerged plants [5-25].

Uptake and Immobilization by the Litter Zone

As discussed previously, some species of wetland vegetation do not translocate nutrients and other compounds to the belowground parts when the stems and leaves die in the fall. The dead plant matter eventually falls to the ground or surface waters and breaks down to form an extensive litter zone in some marshes. Dead, but not decomposed, cordgrass litter in a saltwater marsh was found to be able to adsorb heavy metals directly from the water. Decomposing litter also releases humic acids which act as metal chelators and effectively immobilize these pollutants. Therefore, the organic litter layer in these grass stands acted as a sink accumulating more heavy metals than the living plant mass [5-26, 5-27, 5-28, 5-29].

ADAPTATION MECHANISMS

Vegetative systems, particularly wetlands, are often subjected to widely changing environmental conditions. Mechanisms for coping with salinity and temperature extremes as well as drought and water saturation conditions in the growth medium have evolved in different plant types and species. Through these mechanisms, some plants can become tolerant of pollutant inputs. Some mechanisms may be adapted to a specific pollutant or pollution condition.

Anaerobic Respiration

Wetland systems, by definition, have high water tables at or near the soil surface. These water-saturated soils promote anaerobic conditions at least during some portions of the year and give rise to specific physiological and anatomical adaptations in wetland vegetation. By contrast, in upland systems, prolonged soil saturation and the subsequent development of anaerobic conditions will eliminate most upland vegetation.

In emergent wetland vegetation, the aerenchyma tissue allows for exchange of gases between various plant parts, and is vital to root aeration under anaerobic conditions [5-30]. Laing [5-31, 5-32] has demonstrated that aquatic plants have the ability to respire both aerobically and anaerobically. Of the two types of metabolism, anaerobic respiration can be sustained at a higher rate and for longer periods in aquatic plants but involves a concomitant higher consumption in food material. Other investigations have indicated that the high levels of carbon dioxide which form in the rhizomes of some aquatic plants (aerobically and anaerobically) increase plant photosynthesis [5-33]. However, anaerobic conditions may also inhibit the absorption of certain ions by roots and other vegetative parts [5-34].

Specialized Plant Functions

o Root Interactions

Root aeration during the winter months may provide a buffer from toxic substances present in anaerobic substrates, such as hydrogen sulfide or iron and manganese [5-35]. Seidel [5-36] hypothesized that root excretions may deter root decay induced by bacteria and fungi, and showed that root excretions of some aquatics destroyed disease bacteria while preserving benign populations. An additional function suggested by Seidel, which has particular significance for wastewater application, is the formation of a "protective space" in the root zone allowing benign bacteria to survive during high loadings of heavy metals or other toxic elements, and to subsequently recolonize the area for continued functioning.

o Plant Adaptations

Plant environments which are subject to widely ranging or extreme salinity conditions, such as salt marshes and alkaline salt flats, often have physiological adaptations which allow plants to cope with increased concentrations of salts in the water and soil. These adaptations, in some cases, may also allow the plants to tolerate dissolved contaminants to some unspecified level. Reeds (Phragmites spp.) inhabit areas with a wide range of salinities (0.2 to 40 ppt) [5-37, 5-38]. Structural adaptations for salinity tolerance include starch accumulation, thicker leaves [5-37] and production of different types of fiber [5-39]. The plant also produces more aboveground runners in saline than in freshwater environments, although it is not clear whether this is an avoidance or opportunistic action.

Proline accumulation has been noted in some plants including rushes (Juncus spp.) growing in saline environments. Accumulation of this amino acid may regulate plant osmotic balance with the external medium [5-40].

Reduction of aboveground plant parts to fleshy stems and tubercles, such as in pickleweed (Salicornia spp.), is another adaptation mechanism for dealing with increased salinities and surplus concentrations of materials within plant tissue. Pickleweed can grow in alkaline salt flats where salinities may reach 60 to 80 ppt. Studies in a brackish marsh in California [5-41] have shown the ability of perennial pickleweed to absorb significant quantities of heavy metals beyond soil and water concentrations. The concentrated salts are stored in the more recent plant growth "pickle" parts. At the end of the growing season, Salicornia is able to shed the distal plant parts that, by then, are highly concentrated in salts and are discolored pink or red. By this process, the plant is rid of excess accumulations which eventually recycle back into the wetland system.

- **Transformation to Nontoxic Forms**

Heavy metals and other trace contaminants may be taken up by plants due to the lack of complete specificity of plant absorbance mechanisms. The ability to accept substances that may be potentially toxic to plant functions suggests the existence of a plant compound or compounds which combine with the contaminant to render it nontoxic [5-42]. Thus, some plants may contain chelating agents that would form complexes with specific metal ions, effectively immobilizing the pollutant until the plant matter breaks down.

- **Surplus Accumulation**

The concept of "luxury uptake" of nutrients such as phosphorus and nitrogen has been cited in some studies where plants absorb more nutrients than are needed for plant metabolism. This tends to occur in situations where a particular dissolved substance, required for plant function, is available in higher than normal concentrations and growth conditions are optimal. A University of Michigan study [5-43] showed higher phosphorus concentrations and greater productivity of algae growing near a treated sewage discharge pipe, which can probably be explained by luxury uptake. Most of the surplus phosphorus, however, was apparently immobilized within the algae during growth. Release of the surplus accumulations did not occur until plant decomposition, when the algal phosphorus was converted in the sediments to inorganic phosphorus by bacterial action [5-44]. The action of immobilization and accumulation of surplus substances is similar in concept to the attenuation and accumulation of toxic ions discussed previously. The mechanics and the evolution of these processes are not well understood. However, as discussed later, they do appear to be specific for selected ions, such as heavy metals, in certain plants.

NUTRIENT CYCLING RELATED TO WASTEWATER RENOVATION

Nutrient cycling and potential pathways for some wetland and upland habitats are fairly well known. This knowledge promotes an overall understanding of nutrient movement through an ecosystem and allows predictions for the general outcome of selected parameters under certain conditions. However, quantifying nutrient movements between various components of a cycle is extremely difficult and of limited use, as no two ecosystems, even artificial ones, would be likely to have the same environmental conditions. Although several investigations have yielded percentages or quantities of nutrients retained per area, actual retention effectiveness by compartment and exchange rates have generally not been adequately determined. Thus, in the review of nutrient cycling through vegetative systems, nutrient pathways and their significance in various systems can be discussed, but the nutrient retention or removal potential can only be presented in terms of total systems.

The major nutrients of concern in this review are nitrogen and phosphorus, since lack of these elements is often a limiting factor in the productivity of a vegetation system, and wastewaters are usually rich in these nutrients. Other macronutrients include sulfur, potassium, calcium and magnesium, which are generally adequately supplied by the natural environment. Micronutrients include the trace elements iron, manganese, boron, zinc, copper, molybdenum and chlorine, which are often present in wastewaters beyond plant requirement levels. When accumulated in excess, these micronutrients can cause phytotoxicity (damage to plants). These trace elements are reviewed in a subsequent part of this section.

Nitrogen Cycles in Vegetative Systems

The predominant forms of nitrogen entering wetlands are organic nitrogen, ammonium-nitrogen (NH_4-N) and nitrate-nitrogen (NO_3-N) [5-45]. Nitrogen sources include contributions from ground waters and surface runoff waters, and to some extent from atmospheric nitrogen fixation by some algae and bacteria. Ground-water contributions vary with the extent of ground-water infiltration, nutrient levels in the ground-water, and the extent of nutrient transformation during infiltration (particularly through sediments) [5-46]. Ground-water nitrogen inputs to several marshes studied were generally low, ranging from negligible in a Michigan peatland [5-47] to 0.8 $g/(m^2)(yr)$ in a Wisconsin marsh [5-19] to several mg/l nitrate-nitrogen in another Wisconsin marsh [5-47].

Surface runoff contributions vary as a function of watershed size, and land use practices such as urban or agricultural use. Nitrogen concentrations in rainfall can also be a consideration [5-48]. Typical values for nitrogen in surface runoff as compared to domestic wastewater were presented in Section 3 - Waste Flow Characteristics. Typically, approximately 60 percent of the nitrogen in raw municipal wastewaters is in the ammonium form and 40 percent is organic nitrogen.

Within wetlands, nitrogen fixation has been shown to occur in blue-green algae, bacteria and certain vascular plants, such as Alnus and Myrica [5-15, 5-49, 5-52]. Bristow [5-50] determined that 10 to 20 percent of the nitrogen assimilated by a cattail stand could be derived through fixation. Valiela, Teal and Persson [5-51] found that nitrogen fixation in a coastal salt marsh was decreased with the addition of ammonia.

Nitrogen transformations, particularly within wetlands, are many and varied. Factors contributing to nitrogen forms and concentrations include concentration of inputs, immobilization, ammonification, nitrification, denitrification, vegetative uptake and release, and wetland hydrology. Each of these processes are in turn affected by environmental conditions such as substrate characteristics, pH, BOD, dissolved oxygen (DO), temperature and bacterial action. The nitrogen cycle is complex with many interactions as depicted in Figure 5. In nearly all parts of the cycle, the action of specific bacteria and microorganisms is important.

Hydrologic factors can be influential in the uptake and transformation of various pollutants by wetland vegetation, primarily from the standpoint of water movement and distribution. Nutrient uptake in tidal marshes, for instance, has been found to be most effective where flood waters flow slowly and evenly over the marsh surface [5-55]. During high flow conditions, elevated nutrient levels in outflow waters have been noted and attributed to leaching and flushing of decomposing plant matter [5-47].

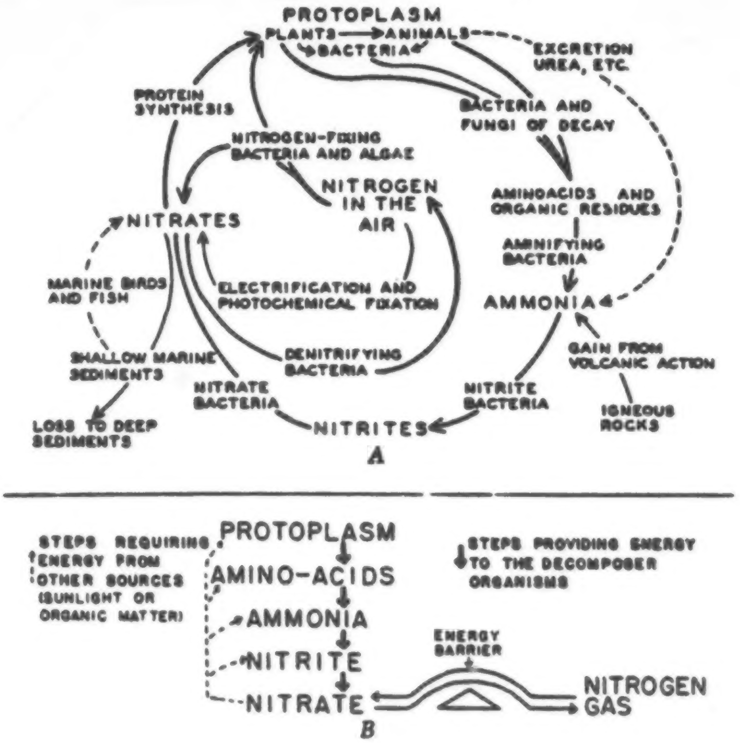
PROTOPLASM
PLANTS
ANIMALS
BACTERIA
EXCRETION UREA, ETC.
PROTEIN SYNTHESIS
BACTERIA AND FUNGI OF DECAY
NITROGEN-FIXING BACTERIA AND ALGAE
NITROGEN IN THE AIR
AMINOACIDS AND ORGANIC RESIDUES
NITRATES
AMINIFYING BACTERIA
MARINE BIRDS AND FISH
ELECTRIFICATION AND PHOTOCHEMICAL FIXATION
AMMONIA
SHALLOW MARINE SEDIMENTS
DENITRIFYING BACTERIA
GAIN FROM VOLCANIC ACTION
NITRATE BACTERIA
NITRITE BACTERIA
IGNEOUS ROCKS
LOSS TO DEEP SEDIMENTS
NITRITES
A
STEPS REQUIRING ENERGY FROM OTHER SOURCES (SUNLIGHT OR ORGANIC MATTER)
PROTOPLASM
AMINO-ACIDS
AMMONIA
NITRITE
NITRATE
STEPS PROVIDING ENERGY TO THE DECOMPOSER ORGANISMS
ENERGY BARRIER
NITROGEN GAS
B

Figure 5. The nitrogen biogeochemical cycle.[a]

[a] Source: reference 5-62.

The depth of water within wetlands may have a pronounced effect on uptake properties of vegetation. For example, some floating macrophytes exhibit the highest uptake rates under shallow water conditions, whereas many emergent species perform best at maximum depths [5-57]. In the study of the Wayzata Wetland, however, no apparent relationship between water levels and phosphorus removal was found [5-58]. Extended ponding of water, beyond natural conditions, is likely to encourage increased biological activity, volunteer vegetation and resultant increases in nutrient removal effectiveness [5-59].

The opportunity for water (and wastewater) contact with wetland vegetation is important to biological removal processes. This is related to detention time, flow distribution and circulation patterns [5-47, 5-55, 5-60]. Well-distributed flow, such as sheet flow, tends to increase the effective area available for biological interaction with water-borne pollutants [5-60].

The seasons clearly are an important factor when considering natural biological systems. With respect to nutrient retention, northern wetlands have generally been found to act as sinks, releasing nutrients during flushing flows in the fall, winter and spring [5-56,5-61].

Best nutrient removal efficiencies are reported in the spring and summer growing season [5-57]. Where winter freezing and snow conditions prevail, the spring snow-melt period tends to be the most critical as far as nutrient discharge is concerned [5-56, 5-61]. Freezing promotes release of nitrogen and phosphorus from plants and soils, making them available for subsequent washout with spring runoff [5-56, 5-61].

- **Ammonification**

Organic nitrogen is broken down, predominantly, to ammonium forms by microorganisms in the substrate and water column of wetlands, although a small fraction may be adsorbed by the sediments [5-49]. This reaction proceeds faster during the growing season and at higher temperatures [5-45]. Ammonification appears to be more significant under anoxic conditions than under aerated conditions [5-53, 5-54]. Lee et al. [5-47] have suggested that this reaction is the most significant rate-controlling reaction in the nitrogen cycle since their marsh studies showed that nitrification proceeded faster than ammonification. If low ammonium levels limit the rate of the nitrogen cycle, then external inputs of wastewater high in ammonium should greatly affect the cycle.

Within wetlands, plant uptake of ammonium is known to occur [5-49, 5-67, 5-68]. Although preferential use of nitrate by phytoplankton and macrophytes has been documented [5-69, 5-70], Patrick and Mahapatra [5-71] and Klopatek [5-16] suggest that ammonium is the most important form of nitrogen available to plants in waterlogged soils. This is due to the rapid breakdown of nitrate (denitrification) under anaerobic conditions making nitrate less available for plant roots.

Fluctuations of ammonium levels appear to be related to the oxygen level of the water column and substrate. Under aerobic conditions, aqueous ammonium levels would be low since either conversion to nitrate (nitrification) [5-49, 5-53, 5-69, 5-72, 5-73, 5-74, 5-76, 5-77, 5-78, 5-79, 5-80] or adsorption within organic soils [5-44, 5-76] would be occurring. Under anoxic conditions, sediment adsorption is lessened [5-82] and can even result in a release of ammonium and other elements to overlying waters [5-82, 5-83]. Lee et al. [5-47] found the highest ammonium levels in a Wisconsin marsh during the spring thaw period, when anoxic waters and sediments were brought to the surface. It was reported that peatlands in Michigan [5-43] and wetlands in Massachusetts [5-85], both receiving wastewater, had highly variable ammonium levels, with the wetland discharge concentrations at times exceeding the inputs. This may be related to changing oxygen conditions favoring microbial activity and ammonification.

- **Nitrification**

The nitrification process refers to the biological conversion (oxidation) of ammonium compounds to the nitrite and nitrite-nitrogen forms. The major agents are the bacteria Nitrosomonas for conversion of ammonium to nitrite, and Nitrobacter for nitrite to nitrate, both in the presence of oxygen. Nitrification occurs almost entirely in the oxidized sediment (water interface and aerated water column in wetlands) where it is estimated that 4.6 mg/l of DO is required to oxidize 1 mg/l of nitrogen [5-79]. Minimal nitrification occurs where the DO is less than 1 to 2 mg/l [5-74], although some activity has been found at DO levels as low as 0.3 mg/l [5-49, 5-53]. Nitrification in the water column is said to depend upon aquatic plants which provide the support structure for slowly reproducing nitrifying bacteria [5-22]. Nitrification in the surface sediments may contribute nitrates to the underlying anaerobic zone, where denitrification proceeds [5-77], and has great significance in wetland nutrient cycles [5-53, 5-54, 5-74, 5-75]. However, bacterial action and oxygen depletion associated with nitrification may add to the BOD of a wetland system, thus making wastewater BOD reduction less efficient. Other work has shown that in some systems, warmer temperatures (above 10 °C) promote nitrification [5-53, 5-74, 5-78] and low pH levels and high salinity inhibit this process [5-49, 5-66, 5-78], particularly in waters below pH 5.0.

- **Volatilization**

Loss of ammonium in wetlands can occur through volatilization across the air-water interface. Studies on pond systems with submerged macrophytes and algal communities in New York [5-72] and Michigan [5-43] found that this process could account for more ammonia removal than is achieved through nitrification, vegetative uptake or sediment adsorption. However, ammonia volatilization is significant only where the pH is greater than 8.0, and may be of limited importance in wetlands and uplands.

o Denitrification

An important process for nitrogen removal is denitrification, where nitrate or nitrite-nitrogen is biochemically reduced to gaseous nitrogen. This process occurs at the anaerobic sediment-water interface and in anaerobic sediment layers. Typical wetland situations provide the appropriate conditions, with anaerobic muds, and abundant organic matter providing energy (in the form of organic carbon) necessary for bacterial metabolic activity. Denitrification rates have been found to decrease with lower temperatures and in acidic conditions [5-49, 5-53]. Shallow water depths, leading to greater mixing of the sediment-water interface by winds and currents may also be a factor, promoting aerobic conditions and decreasing denitrification [5-85].

The site hydrology, particularly of wetlands, is an important seasonal variable affecting aeration, nutrient transport, and possibly, movement of the nitrogen forms. Aeration at the sediment-water interface promotes nitrification in the upper sediment layer. This creates a concentration gradient whereby ammonium in the lower anaerobic layer diffuses into the aerated upper layer and is nitrified. The nitrate produced during this process will either be taken up by plant roots or diffused into the anaerobic zone where denitrification proceeds. These processes have been shown to occur simultaneously in aquatic systems [5-53, 5-74, 5-77]. Denitrification can be the most significant nitrogen removal mechanism in wetlands, where 90 to 95 percent of the nitrate added to wetland soil-water suspensions was reduced to nitrogenous gases in a Massachusetts wetland [5-86]. Nitrogen removal rates of 7.4 and 3.5 kg/(ha)(d) were reported in a salt and freshwater marsh, respectively [5-84].

Wetland environments present ideal conditions for nitrogen sinks. The rate of plant growth is usually faster than the rate of decomposition, due to anaerobic sediment conditions favoring slow decomposition rates. Thus, wetland soils typically contain high organic matter levels, and fall into the peat and muck categories. Abundant organic matter and an aerobic-anaerobic zone at the mud-water interface provide excellent conditions for denitrification [5-64, 5-65]. The potential for nitrogen removal is illustrated by consideration of the area of peat and muck soils in the Sacramento-San Joaquin delta area of California. When these soils are drained, aerobic decomposition of organic matter in the soil above the water table is so rapid that subsidence up to 7.5 cm/yr is observed. The organic nitrogen in the soil is mineralized and converted to nitrate, which appears in the drained soil at high concentrations. A subsidence of 5 cm/yr represents the release of nitrogen in the nitrate form of about 5,050 kg/ha. Despite this enormous input, the drainage waters and ground water in the area maintain low concentrations of nitrate as a result of denitrification in the saturated zone [5-87].

Nitrate removal from floodwater was investigated in relatively undisturbed cores of a fresh water swamp soil and a saltwater marsh soil in Louisiana [5-83]. The latter was more effective in nitrogen removal, with an average rate of 9.2 kg/(ha)(d), while the fresh water swamp soil removed 3.3 kg/(ha)(d). Addition of organic matter to a rice soil was shown in other experiments to have the effect of decreasing the depth of soil through which nitrate had to diffuse before being reduced, and thus, drastically increasing the rate of nitrate removal.

o Uptake by Vegetation

The effect of vegetation upon nutrient cycling in wet and dryland ecosystems is complex and varied. Interactions of the vegetative component with the soil, water, and aerial components must also be considered. Klopatek [5-16] suggests that aquatic vegetation functions as a "nutrient pump," drawing in nutrients from the sediments, the soil-water interface or from the water column, and temporarily immobilizing these nutrients within the plant tissues. This activity increases the nutrient adsorption and exchange capacity in the soil and further reduces nutrient levels in the water.

On the other hand, nitrogen assimilated into wetland vegetation may, eventually, be either translocated to the root components in some species or returned to the litter component during senescence [5-91]. Hydrologic variables can be crucial in affecting the fate of nitrogen in the system. Higher water levels in the fall, winter or spring can create a flushing action leading to high discharges of particulate matter and organic nitrogen from a system.

Tables 18 and 19 present nitrogen removal potentials for a large number of aquatic vegetation species. The list was compiled by investigators at the University of California at Davis [5-42, 5-93, 5-94] where available data were converted or extrapolated to standardized units for comparison. Table 20 presents observed nitrogen uptake rates by aquatic plant species in terms of kg/(ha)(d), kg/(ha)(y) or kg/(ha)(mo). As these studies were often performed for limited time periods, extrapolation of uptake to monthly or annual rates may be unreliable.

Phosphorus Cycles in Vegetative Systems

Phosphorus is commonly reported as total phosphorus and orthophosphate (inorganic phosphorus) forms in most studies, although more detailed analyses may recognize four to eight forms. For the general review of phosphorus cycles, organic and inorganic phosphorus will be the two major forms discussed. In lacustrine systems, the inorganic phosphorus form is most abundant in the sediments [5-204], whereas the organic phase is normally present in the water column [5-52]. In a marsh-wetland system, where sediment-water interactions are characteristic, the inorganic form will probably be more dominant. Phosphorus relationships are generally less complex than nitrogen

TABLE 18. NITROGEN REMOVAL POTENTIAL OF EMERGENT AQUATIC VEGETATION[a]

Species	Removal potential (kg/ha)			Study area and reference
	Aboveground	Belowground	Total biomass	
CATTAILS				
Typha spp.	0.76			Poland; very acid lake (5-98)
	5.71			Poland, acid lake (5-98)
T. glauca	12	207	207	New York; winter biomass in wetlands (5-100)
T. latifolia	220	94.50	315	Wisconsin; marsh (5-16)
	509			Czechoslovakia; fishpond (5-125)
	53.40			S. Carolina; power plant cooling pond (5-134)
	53-310			Wisconsin; marsh (5-19)
T. augustifolia	130-330			Wisconsin (5-19)
	245-467			Czechoslovakia; fishpond (5-125)
T. domingensis	97.8			S. Carolina; power plant cooling pond (5-134)
REEDS				
Phragmites communis	1.37			Poland; acid lake (5-98)
	3.34-15.98			Poland; eutrophic lake (5-98)
	181-409	350-640		Czechoslovakia; fishpond (5-99)
		830		Delaware; salt marsh, maximum biomass (5-105)
	8-11			Poland; lake, three months biomass - unmown/mown (5-110)
	137-409	354-640		Czechoslovakia; fishpond (5-125)
	800			Ukraine; maximum productivity in a eutrophic lake (5-37)
	118-347			Poland; a range of lakes (5-136)
RUSHES				
Juncus gerardii		940		Delaware; salt marsh, maximum biomass (5-105)
		680		Maine; salt marsh, maximum biomass (5-105)
J. roemerianus		1230		Georgia; salt marsh, maximum biomass (5-105)
J. effusus			100-250	S. Carolina; power plant cooling pond (5-127)
SEDGES				
Carex lacustris	32	40	73	New York; wetlands (5-100)
	11	77	88	Wisconsin; marsh (5-16)
C. rostrata	32	40	73	New York; wetlands (5-100)

(continued)

TABLE 18. (continued)

Species	Removal potential (kg/ha)			Study area and reference
	Aboveground	Belowground	Total biomass	
SEDGES (continued)				
C. lanuginosa	9	169	178	New York; wetlands (5-100)
Carex spp.				Michigan; wetlands (5-134)
Cyperus esculentus	10-26			Connecticut; lakes (5-14)
Scirpus fluviatilis	53	154	173	Wisconsin; marsh (5-16)
S. validus	128.40			S. Carolina; power plant cooling pond (5-134)
S. americanus				S. Carolina; power plant cooling pond (5-134)
GRASSES				
Phalaris arundinacea	437			Minnesota; fertilized overland flow (5-109)
Spartina alterniflora		1290		Delaware; marsh, maximum biomass (5-105)
		980		Subtropical; marsh, maximum biomass
	26-177	53-112	90-289	N. Carolina; salt marsh, various fertilization rates (5-114)
	11-134			N. Carolina; salt marsh (5-95)
	18-60			Georgia; salt marsh (5-96)
	20-100			Georgia; salt marsh, domestic wastewater sludge applied (5-96)
S. patens		140		Georgia; salt marsh (5-105)
		270		Delaware; salt marsh (5-105)
		610		Maine; salt marsh (5-105)

a. Source: reference 5-42.

TABLE 19. NITROGEN REMOVAL POTENTIAL OF SUBMERGED VASCULAR PLANTS[a]

Species	Removal potential (kg/ha)		Study area and reference
	Aboveground biomass	Total biomass	
PONDWEED			
Potamogeton spp.		56.50	Minnesota; agricultural drainage ditches (5-166)
P. pectinatus	6		Poland; eutrophic lake, area highly polluted with domestic wastewater (5-197)
P. perfoliatus	6		Poland; eutrophic lake, area highly polluted with domestic wastewater (5-197)
P. pulcher	19.6		New Jersey; artificial ponds, 1 months growth (5-199)
ELODEA			
Elodea canadensis	39.8		New Jersey; artificial ponds, 1 months growth (5-199)
COONTAIL			
Ceratophyllum demersum		4.47	Poland; eutrophic lake (5-198)
	1.34-13.1		Poland; small eutrophic lake (5-200)
WATERMILFOIL			
Myriophyllum excalbescens		89.66	Wisconsin (5-169)
M. spicatum		8.5	Poland; small eutrophic lake (5-168)
	64.94		Wisconsin; shallow, eutrophic, hardwater lake (5-182)
	83.9		New Jersey; artificial ponds, 1 months growth (5-199)
	56.28		Wisconsin; shallow, eutrophic, hardwater lake (5-202)

[a] Source: reference 5-94.

TABLE 20. NITROGEN UPTAKE RATES OF EMERGENT AQUATIC PLANTS[a]

Species	Uptake rates			Study area and reference
	kg/(ha)(d)	kg/(ha)(y)	kg/(ha)(mo)	
CATTAILS				
Typha latifolia	-0.9-1.6			S. Carolina; power plant cooling pond (5-14)
	1.4			Czechoslovakia; marsh growing season (5-97)
		689		Minnesota; marsh maximum aboveground productivity (5-109)
		2630		Temperate climate; total plant mass at maximum possible productivity (5-124)
T. augustifolia			5.9-129.8	England; marsh (5-137)
REEDS AND RUSHES				
Phragmites communis		270 160		The Netherlands wastewater ponds aboveground/ belowground plant mass (5-12)
		437		The Netherlands; maximum aboveground plant mass (5-105)
			0.4-378	England; marsh (5-137)
SEDGES				
Carex lacustris	1			New York; wetlands, growing season (5-101)
Scirpus americanus	-0.32-0.34			S. Carolina; power plant cooling pond (5-101)
S. fluviatilis		208		Wisconsin; marsh (5-102)
S. lacustris		260 320		The Netherlands; wastewater ponds, aboveground/ belowground plant mass (5-12)
GRASSES				
Phalaris arundinacea		124-272		Alberta, Canada; overland flow, wastewater (5-104)
		109-299		W. Canada; overland flow, wastewater (5-90)
Spartina alterniflora		186		Louisiana; salt marsh (5-115)

a. Source: reference 5-42.

because there are fewer chemical transformations of phosphorus in wetlands or other vegetative systems. The two major processes for phosphorus removal from a system are (a) incorporation into the sediments and (b) biological uptake. The phosphorus cycle is shown in Figure 6.

- **Sediment Interactions**

A number of investigators have found that the sediment-water interface of aquatic systems is important in phosphorus cycles [5-49, 5-83, 5-204, 5-205, 5-206, 5-207]. In an aerated water column, inorganic phosphorus may form complexes with various metal compounds such as oxidized forms of iron and manganese [5-49, 5-204, 5-206, 5-208]. These oxidized metals effectively induce phosphorus precipitation and settling on the bottom sediments. The aerobic upper sediment layer can function as a "sink" for these ions. It may even block transport of ions from deeper sediments to the water column [5-82].

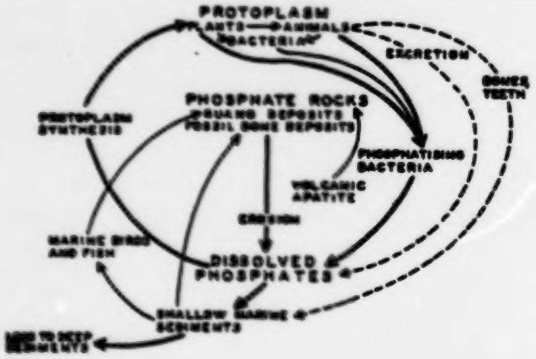
PROTOPLASM
PLANTS
ANIMALS
BACTERIA
EXCRETION
BONES
TEETH
PROTOPLASM SYNTHESIS
PHOSPHATE ROCKS
GUANO DEPOSITS
FOSSIL BONE DEPOSITS
PHOSPHATIZING BACTERIA
VOLCANIC APATITE
EROSION
MARINE BIRDS AND FISH
DISSOLVED PHOSPHATES
SHALLOW MARINE SEDIMENTS
LOST TO DEEP SEDIMENTS

FIGURE 6. Phosphorus biochemical cycle.[a]

a. Source: reference 5-62.

The phosphorus sink functions only under aerobic conditions. Where reducing conditions exist in the sediment, phosphorus and other ions can be released from as deep as 10 cm within the sediment to overlying waters. As mentioned before, an aerated sediment-water interface would form a barrier to the upward migration of phosphorus and other ions. Some interference may occur from oxidized and reduced forms of iron oxides in the sediments, which have different capacities to adsorb and release orthophosphate phosphorus, even under anaerobic conditions [5-206].

o Biological Interactions

Algae and bacteria have been identified as important components in the phosphorus dynamics of some lakes, and may perform an important function in marsh systems as well. Bacterial populations can transform organic phosphorus forms to inorganic phosphorus [5-45, 5-49], and may be associated with orthophosphorus migration to and within sediments through microbial transport [5-49]. Additionally, some genera of bacteria can affect the oxygen status within the sediments, thus promoting phosphorus transformations.

In many studies it was shown that algal uptake of phosphorus was significant [5-47, 5-49, 5-56, 5-70, 5-74, 5-209, 5-210], and that phosphorus was the limiting nutrient for algal growth, or limiting in certain N:P ratios. In a University of Michigan study [5-43], high productivity and phosphorus concentrations were found in algae growing near a stream of secondary-treated sewage efluent. The dynamics are not clear, but it is possible that "luxury uptake" of phosphorus may account for these situations. Luxury uptake of phosphorus has also been observed in activated sludge treatment systems [5-211]. The majority of the phosphorus taken up by the algae is temporarily immobilized within the algae, although small amounts are lost during active growth and recycled to the system. Upon algal decomposition, the organic phosphorus is transformed by bacteria to the inorganic form where it may again be subject to precipitation or adsorption onto the sediments, assimilation by vascular vegetation, or may be transported from the system.

Uptake of phosphorus by vascular plants is the last category of biological interaction in the phosphorus cycle. Prentki [5-19] suggests that vegetation in wetlands may act as a phosphorus "pump," taking phosphorus up from the sediments, temporarily storing it in plant tissue, and then returning it to the sediments through litter decomposition. Active plant uptake of this nutrient probably increases the total phosphorus absorbing capacity of a wetlands system, but the ultimate phosphorus repository will usually be the sediments, reinforcing the function of sediment as a phosphorus sink. Unlike the nitrogen cycle, where significant nitrogen loss in the form of nitrogen gas occurs through the process of denitrification, no similar process exists for phosphorus. The only removal mechanisms are harvesting of aboveground plant parts and transport of phosphorus-containing organic litter (through seasonal hydrologic flushing) out of the system. In this respect, fluctuations in phosphorus levels follow the same pattern as nitrogen levels, with retention during the growing season, and release during high fall, winter or spring flows. Yearly balances in Wisconsin studies [5-89, 5-90, 5-91, 5-92] indicated overall minimal net phosphorus uptake by wetlands systems. Seasonal flushing of phosphorus-containing organic litter apparently accounts for small-to-moderate phosphorus losses but often does not offset net uptake by the system.

In regions that experience warm, dry summers and moderate, wet winters, the characteristic spring release of nutrients in wetlands may be attenuated or may not occur. With mild winters and abundant water supply, some emergent and aquatic plants may continue to grow through the winter; there will be small constant rates of sediment accumulation and flushing, and indistinct spring flushing trends. In fact, significant flushing may occur with the first heavy fall/winter rains that break the summer drought period. The dynamics of this type of marsh system have not been thoroughly studied, and actual release rates are unknown at this time.

- **Uptake Rates by Vegetation**

In general, vegetative systems are not as effective for phosphorus removal as for nitrogen removal. Phosphorus applied to a system eventually is bound to the substrate, and uplands systems which incorporate infiltration through soil layers generally demonstrate the greatest phosphorus removal.

Tables 21 and 22 present phosphorus removal potentials for a large number of aquatic vegetation species. The list was compiled by investigators at the University of California at Davis [5-42], where available data on studies were converted or extrapolated to standardized units for comparison. Table 23 presents observed phosphorus uptake rates by aquatic plant species in terms of kg/(ha)(d), kg/(ha)(y), or kg/(ha)(mo). As these studies were often performed for limited time periods, extrapolation of uptake to monthly or annual rates may be unreliable. As stated earlier, unless the vegetation is harvested and removed from the system, almost all of the phosphorus applied will accumulate and remain within the system, principally in the sediments, or be flushed out [5-224].

UPTAKE AND REMOVAL OF TRACE ELEMENTS

Surface, ground and domestic waters nearly all contain metals or trace elements. Trace elements important in vegetative systems, due to their abundance and/or potential toxicity, are copper, nickel, lead, cadmium and zinc. Some wastewaters have high concentrations of other elements which are usually related to a specific activity or source and must be dealt with in a special manner.

Some of the elements, in trace amounts, are essential plant nutrients although the nutrient requirement and uptake ability vary with different plant species. For example, many soils are low in zinc and enrichment with zinc-laden wastewater can promote plant growth. However, when applied in excess, trace nutrienta can accumulate and pose potential long-term hazards to plant growth and secondary consumers. Copper, zinc, nickel and cadmium are metals that can accumulate in soils and lead to phytotoxicity (damage to plants). Cadmium, and to a lesser

TABLE 21. PHOSPHORUS REMOVAL POTENTIAL OF EMERGENT AQUATIC VEGETATION[a]

Species	Removal potential (kg/ha)			Study area and reference
	Aboveground	Belowground	Total biomass	
CATTAILS				
Typha spp.	0.419-0.481			Poland; acid lakes (5-98)
T. augustifolia	7.9 5.0 4.0			Wisconsin wastewater pond, seasonal totals of biweekly/ monthly/ seasonal harvesting schemes(5-89)
	6-10			Wisconsin; wastewater pond, multiple harvests (5-90)
	31.7			N. Europe; wastewater pond (5-36)
	45-65			Czechoslovakia; fishpond(5-125)
	32-46			Wisconsin; wetlands(5-19)
T. glauca	2.7	39	42	New York; wetlands, winter biomass (5-100)
T. latifolia	43.9	28.6	42.5	Wisconsin, marsh (5-16)
	77			Czechoslovakia; fishpond (5-125)
	6.8-32			Wisconsin; wetlands (5-19)
	24 43			Wisconsin; wetlands, winter/ summer plant mass (5-19)
REEDS				
Phragmites communis	0.126			Poland; acid lake (5-98)
	0.39-1.16			Poland; eutrophic lakes (5-98)
	32-53	38-74		Czechoslovakia; fishpond (5-99)
	0.4			Poland; lake, 3 month growing season (5-110)
	14-63.5			Sweden; lake (5-111)
	62.7			N. Europe; wastewater pond (5-36)
	14-53	38-74		Czechoslovakia; fishpond (5-125)
	80			N. Europe; marshes, maximum productivity (5-37)
	10.6-26.7			Poland; marsh (5-136)
RUSHES				
Juncus effusus			10-30	S. Carolina; power plant cooling pond (5-127)
SEDGES				
Carex spp.	0.96-3.48			Michigan; wetlands (5-34)
Carex lacustris	5	5.5	11	New York; wetlands (5-100)
	2.4	19.7	22	Wisconsin; wetlands (5-16)

(continued)

TABLE 21. (continued)

Species	Removal potential (kg/ha)			Study area and reference
	Aboveground	Belowground	Total biomass	
SEDGES (continued)				
C. rostrata	5	5.5	11	New York; wetlands (5-100)
C. languinosa	1.2	37	38	New York; wetlands(5-100)
C. stricta			59.3	N. Europe; wastewater pond (5-36)
Scirpus fluviatilis	20	32	34	Wisconsin; marsh(5-16)
	11.3			Wisconsin;wastewater pond, total of 4 harvests(5-89, 5-90)
S. validus	35.1-38.3			Wisconsin; wastewater pond, total of 4 harvests (5-89, 5-90)
S. lacustris			67.2	N. Europe; wastewater pond (5-126)
GRASSES				
Phalaris arundinacea	43.71			Minnesota; marsh maximum production (5-109)
	33-56			Pennsylvania; overland flow, wastewater, totals of 3 harvests/season (5-115)
Spartina alterniflora	6			Georgia; salt marsh (5-131)
	1.1-14.9			N. Carolina; salt marsh (5-95)

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a. Source: reference 5-42.

TABLE 22. PHOSPHORUS REMOVAL POTENTIAL OF SUBMERGED VASCULAR PLANTS[a]

Species	Removal potential (kg/ha)		Study area and reference
	Aboveground biomass	Total biomass	
POHDWEED			
Potamogeton spp.		12.9	Minnesota; agricultural drainage ditch(5-166)
P. natans		3.6-11.6	Sweden; small stream polluted with domestic wastewater (5-170)
P. pectinatus	0.6		Poland; lake, site heavily polluted with domestic wastewater(5-197)
P. perfoliatus	0.6		Poland; lake, site heavily polluted with domestic wastewater(5-197)
P. pulcher	3.5		New Jersey; artificial ponds, 1 months growth (5-199)
ELODEA			
Elodea canadensis		0.03-0.93	Poland; eutrophic lake(5-168)
	12.1		New Jersey; artificial ponds, 1 months growth (5-199)
COONTAIL			
Ceratophyllum demersum		0.58-0.99	Poland; eutrophic lake (5-168)
	0.62-6.08		Poland; eutrophic lake (5-200)
WATERMILFOIL			
Myriophyllum excalbescens		8.97	Wisconsin (5-169)
M. spicatum		0.015-0.078	Poland; eutrophic lake(5-168)
	12.55		Wisconsin; shallow, eutrophic, hardwater lake(5-171)
	2.90-17.70		Wisconsin; lake(5-198)
	2.20-12.30		S. Carolina; power plant cooling pond(5-198)
	20.3		New Jersey; artificial ponds, 1 months growth (5-199)
	1.13-5.06		Wisconsin; highly alkaline lake(5-201)
	12.99		Wisconsin; shallow, eutrophic, hardwater lake(5-202)

[a] Source: reference 5-94.

TABLE 23. PHOSPHORUS UPTAKE RATES OF EMERGENT AQUATIC PLANTS[a]

Species	Uptake rates			Study area and reference
	kg/(ha)(d)	kg/(ha)(y)	kg/(ha)(mo)	
CATTAILS				
Typha augustifolia			0.7-32.1	Temperate marsh (5-137)
T. latifolia	-0.03-0.19			S. Carolina; power plant cooling pond (5-13)
	0.08			Czechoslovakia; marsh, growing season (5-97)
	0.95	31		Wisconsin; marsh, growing season total biomass (5-19)
		74		Minnesota; maximum productivity (5-109)
		403		Temperate climate; maximum productivity, total biomass (5-124)
REEDS AND RUSHES				
Phragmites communis		35 20		Temperate climate; wastewater ponds aboveground/ belowground (5-12)
			0.13-19.6	England; marsh (5-137)
SEDGES				
Carex lacustris	0.06			New York; wetlands, growing season (5-101)
Scirpus americanus	-0.04-0.05			S. Carolina; power plant cooling pond (5-13)
S. lacustris		50 55		The Netherlands; wastewater ponds aboveground/ belowground (5-12)
S. fluviatilis		53.3		Wisconsin; marsh (5-16)
GRASSES				
Phalaris arundinacea		70.14		Temperate climate; wastewater irrigation (5-116)
Spartina alterniflora	0.06			Georgia; salt marsh (5-131)

a. Source: reference 5-42.

extent copper, can become hazards at high concentrations to secondary consumers of plants enriched with these elements.

Physical and chemical interactions that can remove trace elements from waters were discussed in Section 4. In most vegetative systems, the physiochemical interactions of pollutants and sediments are the primary removal mechanisms. Biochemical interactions are secondary mechanisms for incorporation of trace elements into a system, and can provide additional capacity for pollutant removal in a system [5-113, 5-164]. Further, since increasing the organic content of sediment improves its storage capacity of heavy metals, the natural replenishment of organic material in a wetland aids the physiochemical process [5-224].

Emergent Aquatic Vegetation

Heavy metals are adsorbed by aquatic vegetation primarily via the roots from the sediment, although leaf absorption of heavy metals from waters also occurs to a lesser extent. As discussed under physical and chemical sediment interactions, pollutant concentrations are usually highest in the upper sediment layers, specifically the top few centimeters. Therefore, rooted emergent vegetation, particularly shallow-rooted species and types with creeping rhizomes near the sediment surface, is especially susceptible to heavy metal uptake.

Tables 24 and 25 present heavy metal removal potentials for several types of emergent aquatic vegetation, based on research where uptake was reported in terms of specific volume or area. Characteristics of several vegetation types are discussed below.

o Cattails

Maximum uptake of heavy metals in aboveground tissues occurs during the growing season, prior to the production of peak aboveground biomass. During the dormant season, heavy metals are not translocated back to the rhizomes, but remain in the stem and leaf parts. Cattails, particularly Typhus latifolia, may be useful in removing copper and moderate amounts of manganese and zinc, as shown in Tables 24 and 25. The iron uptake rate in terms of total plant mass has been measured at 23 kg/(ha)(y), assuming maximum production [5-124].

o Reeds and Rushes

The available literature reports removal rates for reeds only. The common reed Phragmites communis exhibits a seasonal uptake and storage pattern, where metals are taken up during the spring and summer, with the peak uptake occuring in late summer. By the fall, the seeds show concentrations two to four times greater than the leaves and stems. In one case, the copper concentration in the seeds was found to be 30 times greater than in leaf tissue. During the fall, metal uptake levels off and decreases through the winter, with a concomitant rise in

TABLE 24. HEAVY METAL REMOVAL POTENTIALS OF EMERGENT AQUATIC PLANTS (Cd, Co, Cu, Fe, Hg)[a]

Species	Removal potential (kg/ha)					Study area and reference
	Cd	Co	Cu	Fe	Hg	
CATTAILS						
Typha augustifolia		0.006	0.068	15.80		W. Europe; wastewater ponds (5-36)
T. latifolia		0.010	0.360			Ukraine; reservoir (5-126)
REEDS AND RUSHES						
Phragmites communis		0.028	0.188	41.20		W. Europe; wastewater ponds (5-36)
		0.004	0.350			Ukraine; reservoir(5-126)
SEDGES						
Carex stricta		0.020	0.152	103.40		W. Europe; wastewater pond (5-36)
Scirpus lacustris		0.023	0.161	26.20		W. Europe; wastewater pond (5-36)
GRASSES						
Phararis arundinacea	0.001-0.005					Pennsylvania; overland wastewater disposal; varied rates (5-22)
Spartina alterniflora				0.19-9.90		N. Carolina; salt marsh(5-99)
				3.84		N. Carolina; salt marsh(5-133)
	0.004	0.026		5.25	0.001	S. Carolina; salt marsh(5-28)
S. alterniflora and S.patens	0.0003-0.0050					Massachusetts; salt marsh, wastewater sludge applied at various rates to a mixed stand (5-10, 5-26)
OTHERS						
Justica americana			0.30-0.80	9.8-38		Alabama; lake (5-135)
Salicornia pacifica	0.08-0.38		0.42-1.58			California; brackish marsh receiving urban runoff (5-41)

a. Source: references 5-41, 5-42.

TABLE 25. HEAVY METAL REMOVAL POTENTIALS OF EMERGENT AQUATIC PLANTS (Mn, Mo, Ni, Pb, Zn)[a]

Species	Removal potential (kg/ha)					Study area and reference
	Mn	Mo	Ni	Pb	Zn	
CATTAILS						
Typha augustifolia	11.22	0.004	0.027		0.629	W. Europe; wastewater ponds (5-36)
T. latifolia	13.66				0.600	Ukraine; reservoir (5-126)
REEDS AND RUSHES						
Phragmites communis	7.44	0.012	0.068		1.658	W. Europe; wastewater pond (5-36)
	15.60		0.068		0.500	Ukraine; reservoir (5-126)
SEDGES						
Carex stricta	25.38	0.008	0.067		1.714	W. Europe; wastewater pond (5-36)
Scirpus lacustris	40.32	0.018	0.058		1.680	W. Europe, wastewater pond (5-36)
GRASSES						
Phalaris arundinacea				0.106-0.437		Pennsylvania; overland wastewater disposal; varied rates (5-22)
Spartina alterniflora	0.02-0.42					N. Carolina; salt marsh(5-99)
S. alterniflora				0.028-0.116		Massachusetts; salt marsh, wastewater sludge applied at various rates (5-129)
	0.32				0.06	N. Carolina; salt marsh(5-133)
	0.35					S. Carolina; salt marsh(5-28)
S. alterniflora *S. patens*					0.048-0.301	Massachusetts; salt marsh, wastewater sludge applied to a mixed stand at various rates (5-10, 5-26)
OTHERS						
Justicia americana	1.3-2.5				2.6-5.8	Alabama; lake (5-135)
Salicornia pacifica				0.026-1.01	0.43-0.68	California; brackish marsh receiving urban runoff (5-41)

a. Source: references 5-45, 5-42.

concentration in underground parts. By late winter, metal accumulation is entirely within the underground parts with 2 to 10 percent appearing in the shoots [5-24]. Except for the late fall and winter when reed metabolism appears to be geared toward seed production, root concentrations are always greater than leaf or stem concentrations. Reeds may be useful in removing significant amounts of copper and iron [5-24, 5-36, 5-126] and moderate amounts of cobalt and molybdenum [5-36].

o Sedges

In laboratory studies, mercury uptake by a bulrush, Scirpus cyperinus was primarily through surface adsorption by submerged shoots instead of uptake by roots from sediments. Mercury adsorption through the exposed tissues increased with increasing aqueous mercuric chloride concentrations [5-128]. S. lacustris showed significant removal potentials for zinc (.058 kg/ha) in a wastewater pond [5-36]. Also in a wastewater pond, Carex stricta has high removal potentials for iron (103.4 kg/ha), nickel (0.67 kg/ha) and manganese (26.38 kg/ha) [5-36].

o Grasses

Reed canarygrass (Phalaris arundinacea) accumulates metals in proportion to soil and water concentrations [5-22, 5-23]. Lead translocated from roots to aboveground parts does not return to the roots during the winter [5-23]. Assuming that this is true for other metals, then pollutant removal from the system would occur only through external harvesting. The removal potential of reed canarygrass ranges from 0.001 kg/ha (cadmium) to 0.69 kg/ha (copper).

Cordgrass (Spartina spp.) is similar to canarygrass in that translocation of accumulated heavy metals to belowground parts does not occur during leaf senescence. However, the role of cordgrass in an estuarine-salt marsh ecosystem leads to other interactions that are more important in heavy metals removal. Undecomposed cordgrass litter above the sediment layer absorbs metals directly from the water, while decomposing litter releases humic acids, which act as metal chelators. Thus, the cordgrass litter compartment can actually accumulate more heavy metals than the living plant mass and act as a sink for heavy metals. The living cordgrass system, in comparison to other freshwater emergent vegetation, generally shows lower heavy metals removal potentials, as shown in Tables 24 and 25.

Anaerobic conditions in a salt marsh, where mercury has accumulated in the sediments as insoluble sulfides, can lead to the production of toxic methyl mercury through bacterial methylation. Whereas mercury would normally be absorbed by roots from the sediments, methyl mercury can be absorbed by leaf and stem tissue too, accumulating tissue concentrations higher than concentrations in the surrounding soil [5-29, 5-130].

- **Other Types**

In the water willow (Justicia americana), maximum uptake of heavy metals in aboveground tissues occurs prior to the production of peak aboveground biomass. Little is known about the dormancy pattern. J. americana has significant removal potentials for zinc (2.6 to 5.8 kg/ha), manganese (1.3 to 2.5 kg/ha) and copper (0.30 to 0.80 kg/ha) [5-135].

The perennial pickleweed (Salicornia pacifica) has significant removal potentials for cadmium (0.016 to 0.024 kg/ha) [5-41]. Uptake probably occurs from the soil to the roots and aboveground parts, although direct absorption through stem/leaf tissue can also occur. Data are scarce on the overwintering patterns and heavy metals cycling through the system.

Floating Aquatic Vegetation

Root characteristics are probably most important in determining the heavy metal removal potential of floating vegetation. Roots may be simple and sparse as in duckweed or fibrous and matted as in water hyacinth. Another consideration is water depth and root length. Where roots are able to extend and grow in sediment, the removal potentials may vary. Sediment-rooted submerged vegetation generally absorbs fewer minerals from the water than those rooted in a nonsoil medium or with entirely aquatic roots [5-22]. This may be the case in floating vegetation as well.

- **Water Hyacinth**

The intricate root system of water hyacinth mats has a strong affinity for suspended particulates. Heavy metals associated with particulate matter are probably removed through the root filtering mechanism. Measurements have been made of decreases in pollutant concentrations accompanying hyacinth growth [5-123] and of various metals recovered in hyacinth plants [5-138, 5-141, 5-150, 5-152, 5-155, 5-156, 5-158]. The highest metal concentrations occur in the roots [5-148, 5-150, 5-158] with some translocation to upper plant parts [5-150, 5-152, 5-155, 5-158]. However, the significance of heavy metal removal through physical and chemical processes associated with the roots, as compared to biological uptake, has not been determined. Water hyacinth may be useful in removing iron, mercury, arsenic, chromium, copper, lead and nickel where biomagnification factors of plant tissue accumulations can be 50 to 200 times the water concentration. For iron and mercury, the biomagnification factor can rise to 6,600 times and almost 700 times, respectively. Concentrations of heavy metals in water hyacinth reported in experimental studies are given in Table 26.

- **Duckweed**

The high productivity of duckweed can lead to active uptake and accumulation of trace metals beyond ambient water concentrations [5-139,

TABLE 26. HEAVY METAL CONTENT OF FLOATING AQUATIC VEGETATION[a]

Species	Dry weight, mg/g												Comments and reference
	Al	Cd	Cr	Co	Cu	Fe	Pb	Hg	Ni	Ag	Sr	Zn	
Water hyacinth		300					3200						96 hrs. in 0.1 ppm Cd and 10 ppm Pb solutions (5-138)
				0.5-2.9	0.4-2.6	15-70							Plants of different sizes from natural populations (5-151)
		200-670							300-500				24 hrs. in 0.6-2 ppm solutions (5-153)
		<.001		<0.007	<0.01		0.063	<0.001	<0.05	<0.02	<0.01	0.58	2 wks. in sewage effluent; measurements made on roots only (5-154)
				89-568						140-650	102-544		24 hrs. in 0.6-2.4 ppm solutions. Uptake measured as difference in metal content of substrate (5-155)
		2-164	4-286		15-570		25-257			4-77			6 wks. in chemical waste system. Lowest concentrations in leaves and stems; highest in roots (5-157)
		<0.2-281											24 hrs. in 0.001-0.1 ppm solutions. Lowest concentrations in plant tops; highest in roots (5-158)
Duckweed					3-110								Naturally-occurring population, England (5-159)
	468	17.4	65	19.2	101.1	13.9		5.6			679	58	Plants growing in drainage system of coal-fueled power plant, S. Carolina (5-162)
	1980			26	79				1840				American River, California (5-163)

[a] Source: reference 5-93.

5-160, 5-163, 5-165]. In Spirodela polyrhiza, removal potentials for manganese and cobalt were greater than those of other emergent or floating aquatic vegetation [5-164]. Field studies on Lemna minor revealed biomagnification factors of 20,000 to 100,000 times ambient water concentrations for cobalt, copper, nickel and titanium; and 300,000 to 660,000 times ambient water concentrations for iron, manganese and aluminum [5-163]. Since duckweed normally grows without sediment contact, uptake must be directly from the water. It is possible that the continual decay process, associated with plants that have high productivity, releases these trace elements continuously to the upper waste layers, so that they are recycled to new plant growth [5-163]. Concentrations of heavy metals in duckweed, from experimental studies, are given in Table 26.

Submerged Aquatic Vegetation

Submerged vascular plants cycle trace elements in a similar manner to emergent vegetation. Heavy metals are absorbed through the roots from the sediments, and are subsequently translocated to other plant parts. As the plant dies and decomposes, plant elements are recycled back to the water. Submerged plants are generally shallow-rooted, and are thus exposed to the higher concentrations of heavy metals that settle out within the upper sediment layer [5-196].

Submerged plants are generally found in alkaline, hard waters (pH 7 to 10) where they have the ability to utilize bicarbonate as a carbon source when the high pH decreases the amount of free carbon in the water. Carbonate deposition on plant leaves and stems is a side effect of bicarbonate or carbon dioxide absorption by submerged plants in alkaline calcareous waters. Marl, as these deposits are called, is formed by excess bicarbonate within the plant reacting with hydroxyl ions at the leaf surfaces, and precipitating with calcium onto plant surfaces. Marl deposit formation can be an important mechanism for capturing potentially hazardous ions that coprecipitate with calcium, such as strontium.

A comparison of heavy metal removal potentials of several types of submerged vegetation is presented in Table 27. Generally, submerged plants have a lower plant mass than emergent or floating vegetation, and thus, lower removal potentials for heavy metals. This becomes apparent when comparing data on submerged plants with the reported data on other vegetation types. However, the removal potentials of pondweeds for cobalt and copper can equal the potentials for reeds and cattails. Several submerged plant species also exhibit tolerances for specific metals as discussed below.

- **Pondweed**

Potamogeton sp. tend to concentrate heavy metals in the roots and rhizomes, where the primary contact occurs. Zinc, however, accumulates in the stems and leaves [5-178]. P. pectinatus readily translocates

TABLE 27. HEAVY METAL REMOVAL POTENTIALS OF SUBMERGED VASCULAR PLANTS[a]

Species	Removal potential (kg/ha)							Comments and reference
	Al	Cd	Co	Cu	Fe	Mn	Zn	
PONDWEED								
Potamogeton crispus		0.001-0.114						Indiana; shallow eutrophic lake; soil Cd concentrations 0.66-44.8 ppm (5-175)
P. lucens			0.024	0.087		4.250	0.400	Ukraine; shallow eutrophic reservoir. (5-126)
P. pectinatus			0.010	0.030		2.500	0.210	Ukraine; shallow eutrophic reservoir. (5-126)
					0.140	0.040	0.001	Poland; eutrophic lake, site heavily polluted with domestic wastewater, reduced growth. (5-197)
P. perfoliatus					0.140	0.040	0.001	Poland; eutrophic lake, site heavily polluted with domestic wastewater, reduced growth. (5-197)
WATERMILFOIL								
Myriophyllum spicatum	0.109			0.007	0.130	0.109	0.002	Wisconsin; shallow eutrophic hardwater lake. (5-171)

a. Source: reference 5-94.

lead from roots to shoots, where it accumulates without adverse effects [5-181]. Incorporation of lead into the cell walls apparently changes it into a form that is nontoxic to the plant [5-179]. In another study, *P. crispus* growing in a river delta was found to have higher concentrations of copper, manganese and chromium than the common reed (*Phragmites communis*), although levels in *Elodea* were even higher [5-24].

Several studies have been done on strontium and calcium uptake in pondweeds. The two elements are interchangeable in plant metabolism, although the affinity for calcium is greater in most plants [5-176, 5-177] Strontium, however, can be incorporated with calcium into the carbonate or marl deposits as the plant rids its system of excess carbonate. By this mechanism, strontium-90, an environmentally hazardous radionuclide, can be sequestered into a sparingly soluble plant deposit [5-186].

o Elodea

Elodea is a rapid-growing submerged plant that can grow rooted in sediment, or become fragmented and free-growing in water. Heavy metals can be absorbed readily through both roots and shoots, with sediment-rooted plants concentrating elements in the roots. Heavy metal contents of plants are proportional to environmental concentrations [5-24, 5-183, 5-185, 5-187, 5-188]. *Elodea* can cycle elements rapidly through an aquatic system, taking up heavy metals from soil and water, retaining a portion within the plant, and releasing 60 to 70 percent of the initial amount absorbed, back into the water [5-183, 5-184, 5-185, 5-186, 5-187, 5-188].

Elodea can take up mercury in direct proportion to water concentrations at rates of 1 to 33 mg/(g dry wt)(d), accumulating up to 1,000 mg/g dry wt in laboratory studies [5-189].

Inorganic mercury and organic mercury (in the form of methyl mercury) are taken up without specific preference. In a study on *Elodea densa*, plants growing in an inert nonsoil medium (glass beads) accumulated significantly more mercury (up to 32 percent) than those rooted in soil. The soil apparently contributed enough plant nutrients so that mineral uptake (including mercury) from the water was reduced. Soil in which *Elodea* had grown was also found to accumulate about twice as much mercury as plant-free soil due possibly to: (a) increased surface area of soil from disturbance by plant roots leading to increased sediment-mercury interactions; or (b) deposition of mercury-containing suspended material, algae and dead leaves on the soil surface [5-189].

Although removal potentials of *Elodea* have not been documented in terms of kg/ha, various studies have shown that *Elodea* can be more effective than other submerged plants and more effective than emergents, such as the common reed, in removing copper, manganese and chromium from water [5-182, 5-189, 5-190].

- **Coontail**

Ceratophyllum sp. absorbs heavy metals primarily through the foliage, as opposed to roots in the sediment [5-191]. Under experimental conditions, uptake is highest during rapid growth periods, reaching a maximum at 6 to 9 weeks and decreasing by the twelfth week [5-193]. In laboratory studies, Ceratophyllum demersum absorbs both ethyl mercury and methyl mercury and is able to convert both into inorganic mercury [5-192].

- **Watermilfoil**

Absorption of heavy metals through the foliage or roots of watermilfoil has not been verified. As the plants are relatively fragile and reproduce mainly by fragmentation, as do Elodea and coontail, they probably exhibit rapid seasonal growth and uptake of nutrients and trace elements through the foliage. Limited data are available on heavy metal uptake rates by watermilfoil. In field studies comparing pollutant concentrations in watermilfoil to concentrations in the emergent plants Glycera sp. and Chara sp., watermilfoil had higher accumulations [5-194, 5-195, 5-196]. However, the overall removal is probably lower than removals by the emergent cattails and reeds.

UPTAKE AND REMOVAL OF OTHER POLLUTANTS

Other pollutants of environmental concern that may be present in wastewaters include the trace element boron, and organic compounds such as petroleum products and pesticides. The ability to accumulate these pollutants has been studied for a few plant species only and the data are generally insufficient to make comparisons of removal effectiveness between vegetation types.

Boron Uptake

Boron is absorbed in plant systems by the same mechanisms discussed for heavy metals. Primary uptake occurs through the roots, and accumulation in aboveground parts reaches a peak when maximum plant biomass is attained. Grasses (monocotyledons) have a lower boron requirement than dicotyledons, and probably have lower removal potentials for boron.

- **Emergent Aquatic Plants**

The boron removal potentials for emergent plants are all based on studies of wastewater or cooling ponds in temperate climates. Cattails and reeds have similar removal potentials of 0.030 to 0.352 kg/ha [5-117, 5-118] and 0.37 kg/ha [5-36], respectively. Bulrushes range from 0.011 kg/ha in a South Carolina cooling pond [5-117] to 0.496 kg/ha in a wastewater pond [5-118]. Carex stricta exhibited the highest removal potential among emergent plants of 0.582 kg/ha [5-36]. Boron removal potentials of 0.007 to 0.029 kg/ha were also found in various species of water lilies [5-119].

- **Floating Aquatic Plants**

Boron uptake has been studied in only one floating plant, Lemna minor (duckweed). Calculated uptake rates for boron in experimental aqueous concentrations of 0.01 to 1.01 mg/l ranged from 159 to 160 mg/(g ash-free dry wt)(wk). Boron concentrations in Lemna were 4 to 70 times greater than concentrations in other submerged aquatic plants collected from the same wastewater ponds [5-149].

- **Submerged Aquatic Plants**

Few submerged plants have been studied for boron removal potential. The highest reported potential is 0.11 to 0.23 kg/ha for a community of pondweed, elodea and coontail, harvested from a Michigan wastewater pond [5-167]. Generally, submerged plants appear to have a lower boron removal potential than emergent or floating plants. Potamageton diversifolius, growing in boron-containing waters of 0.1 to 5.2 ppb, showed a concentration factor of 10 to 50 times in the plant tissues [5-117]. Boron removal potentials of 0.007 to 0.083 kg/ha during the growing season have been reported for coontail (Ceratophyllum demersum) [5-172]. Myriophyllum heterophyllum is reported to be able to attain boron concentrations 21 to 100 times natural water levels [5-117]. A removal potential of .007 kg/ha was reported for Myriophyllum spicatum in a Wisconsin Lake [5-171].

Organic Compounds

The organic matter represented by BOD in municipal wastewater effluent and stormwater will, initially, be removed by sedimentation or stay in solution or colloidal suspension in a wetland. The settled organic material will be microbially digested in the bottom sediment. High organic loads to the wetland may induce anaerobic conditions in what might otherwise be aerobic sediments.

The soluble and colloidal organic matter will be metabolized by microorganisms attached to plant roots and stems, and to a much lesser extent, by free floating microorganisms in the water column.

Organic-nitrogen derived from metabolic wastes and dead organisms is converted to elemental nitrogen (or mineralized) by microorganisms in soils and waters. Many synthetic organic chemicals are also broken down by microorganisms. The mineralization process frees carbon and releases energy for microbial biosynthesis. Detoxification is a common result of mineralization [5-88, 5-102].

A microbial conversion different from mineralization can also take place. Compounds are acted upon biologically in soils and waters, but not all of the products resulting can be used as nutrient or energy sources. The microbes will utilize those products compatible with energy and nutrient needs, while the unusable by-products gradually accumulate [5-173, 5-174]. The products are generally nontoxic or of low toxicity, although they may be subject to biomagnification through plant uptake and the food chain. Some products may be toxic only to specific organisms. In rare but significant instances, the products can be highly toxic, such as methylmercury produced from the methylation of mercury by bacteria.

Organic material in a wetland discharge, will typically consist of [5-224]:

- o extracellular organic compounds leached during the growing season;
- o organic compounds leached from decaying vegetation;
- o algae and microbes suspended in the water column;
- o biorefractory organics in the influent.

Petroleum products, phenols and phenol derivatives have been studied in conjunction with oil spills and urban runoff entering vegetative systems. Other organic compounds that have been studied on an intermittent basis include growth substances, herbicides and insecticides. Uptake of organic compounds by submerged plants has not been studied [5-42].

o Emergent Aquatic Plants

Petroleum products are not actively taken up by emergent plants. Incidences of oil spills which spread to wetland areas generally did not cause noticeable damage to the vegetation. However, the emergent vegetation is able to act as a biological filter, promoting the proper conditions for breakdown of hydrocarbons, while the hydrocarbons are actually decomposed (oxidized) by bacteria [5-103, 5-108, 5-112, 5-118, 5-132, 5-180, 5-203]. Plant stems and leaves provide bacterial attachment sites, and thus, increased contact/treatment areas for hydrocarbons. The presence of cattails and bulrushes has been reported to increase the hydrocarbon decomposition rate to as much as 7 times the rate measured in the absence of emergent vegetation. [5-120, 5-121]. While reeds, rushes and grasses have not been studied, a similar bacterial mechanism may also be operative. In European wastewater treatment ponds, phenol uptake rates by bulrushes is 4 mg/(100 g plant)(d). Phenol derivatives included p-cresol, xylene, pyrocatechol, resorcinol, pyrogallol, pyridine and hydroquinone [5-121].

o Floating Aquatic Plants

Reduction of phenol concentration in solution from 25 to 100 mg/l to less than 0.5 mg/l in 72 hours appeared to be related to the presence of water hyacinth [5-143]. However, phenol was not recovered in the plant tissue and the phenol removal or breakdown mechanism is uncertain. In another study, water hyacinth absorbed 1.30 to 2.5 mg toxaphene/plant from a 2 mg/l solution. Toxaphene, an organic insecticide, appeared to be taken up most rapidly in the first 48 hours [5-142].

The uptake and accumulation of growth substances such as 2,3,5-trichlorobenzoic acid (TBA), indoleacetic acid (IAA), phenoxyacetic acid and hexose have been studied for two species of duckweed: *Lemna minor* and *Lemna gibba* [5-144, 5-145, 5-146]. Temperature, pH, growth substance concentration, light, and presence of chlorine affect the accumulation rates in different plant parts [5-140, 5-144, 5-146].

Uptake of Biocides

Insecticides and herbicides are applied to aquatic systems to reduce insect vectors and nuisance plants and algae. Accumulation of these biocides in plant tissue affects considerations for the ultimate use or disposal of plant material. This can be important in the food chain where secondary consumers can concentrate these biocides and experience toxic effects. In addition to physical and chemical removal of the biocides, the plants and highly varied microbial populations of a wetland will metabolize many organic biocides.

o Emergent Aquatic Plants

Biocide uptake has been studied in water lilies and jointgrass. The pesticides hexachlorocyclohexane (HCCH) and DDT accummulated 10 to 12 times ambient water levels in the water lily *Nymphaea alba* [5-122]. The insecticide mevinphos was absorbed at the rate of 7 ppm/d by *Nymphaea odorata* and *Paspalum distichum* (jointgrass) [5-123]. The submerged rush *Juncus repens* had no effect on mevinphos removal.

o Floating Aquatic Plants

Water hyacinth can achieve rapid removal of diphenamid, a weed control agent, through degradation and release of metabolics back into the water [5-147]. Applications of the herbicide copper sulfate pentahydrate over 2 mg/l inhibited water hyacinth growth and led to increased copper concentrations in the roots [5-148].

o Submerged Aquatic Plants

At low biocide concentrations in the aquatic medium, submerged plants appear to be able to accumulate and metabolize biocides without showing toxic effects. Uptake of organo-chlorine insecticides is dependent upon plant lipid content and contact surface area available for absorption. Submerged plants, such as pondweed allow contact with the entire plant surface. *Potamogeton pectinatus* accumulated DDT and HCCH at average levels of 3.80 and 0.94 mg/kg dry wt [5-212]. DDT uptake comparisons have also been performed on different pondweed species [5-213].

Removal of the pesticide pollutants dichlobenil, diphenamid and amitrole were measured on *Elodea canadensis*, *Potamogeton diversifolius* and *Myriophyllum spicatum*. All plants were affected by dichlobenil concentrations of 0.17 mg/l and took up small amounts of diphenamid. Only *Elodea* accumulated amitrole [5-217]. *Myriophyllum brasiliense* can degrade diphenamid to a relatively nontoxic monomethyl derivative [5-214]. *M. brasiliense* also absorbs the herbicide simazine through the roots [5-215].

Bacteria and Viruses

Little information is available on the removal of pathogens in wetland systems. Bacteria and viruses are subject to sedimentation, exposure to ultraviolet radiation, and attack by chemicals. Additionally, natural predation by other organisms will occur in a wetland. Pathogens which rely on a specific host may also die off. However, reduction rates and the safety from transmission of diseases to either humans, animals or even plants have not been established.

Other Water Quality Parameters

Within the water column, variations in pH, salinity or electrical conductivity, dissolved oxygen and water color are most evident under different wetland conditions.

o pH Levels

Factors that have been observed to influence pH levels in wetlands water quality are (a) the nature and source of wetland inflow [5-216], and (b) high photosynthetic activity within the wetland [5-47].

Typical surface waters entering wetlands exhibit neutral pH values between 6.5 and 7.8. Variations in pH occur where upstream sources may include acid mine drainage, municipal, commercial or industrial discharges, and specific geologic or mineralogic soils.

In Verry's study of Minnesota perched (ombrotrophic) and ground-water (minerotrophic) wetlands [5-216], striking differences were found in pH. The perched bogs, with no ground-water inflow, had pH values of 3.1 to 4.2, totally independent of streamflow. The suspected sources of hydrogen ions were (a) dissociation of sulfuric acid derived from hydrogen sulfide and (b) cation exchange in the cell walls of *Sphagnum* plants. In contrast, the minerotrophic wetland was neutralized (pH 6.5) by ground-water inflow containing calcium, and presumably, bicarbonate ions leached from upland mineral soils.

Less drastic variations in pH (7.1 to 7.5) were noted by Lee et al., [5-47] in the study of Wisconsin marshes. High photosynthetic activity was noted as the cause of major diurnal variations in dissolved oxygen and concomitant fluctuations in pH. Winter-time decreases in pH were also noted, and found to be associated with winter-time dissolved oxygen depletion and production of sulfide.

Studies of the Holland Marsh [5-218] and Jewell Pond [5-217] also indicated relatively low pH levels of 6.2 to 6.8 and 6.0 to 7.0, respectively. Hall et al., [5-217] also found low buffering capacity and tendency for pH to decrease from water surface down into the bottom muds of the Jewell Pond wetlands.

Levels of pH play an important role in the chemistry and biochemistry of wetlands in that they may influence the availability of nutrients to plants and also affect precipitation reactions involving metal ions [5-216].

- **Dissolved Oxygen**

Large amounts of photosynthetic activity and accompanying respiration by macrophytes, epiphytes and benthic algae were found by Lee et al., [5-47] to lead to diurnal variations in DO concentrations in waters discharged from marshes. Winter ranges of 6 to 9.5 mg/l were observed. During the summer growing season much greater variation (0 to 8 mg/l) was noted. In one particular marsh, thermal stratification during the winter resulted in depletion of DO and production of sulfide. Throughout the investigation DO was almost always at less than 50 percent of saturation.

Reaeration of natural waters has been studied by many researchers primarily because of interest in self-purification aspects of streams. The rate of reaeration of flowing water has been found to vary directly with flow velocity and channel roughness and inversely with water depth [5-219, 5-220]. The combined effects of flow resistance from vegetative cover and regular water movement are generally sufficient to maintain high DO levels in shallow water wetland areas. In a carefully controlled study of bog waters, Sparling [5-221] found that at velocities greater than 1 cm/s, waters were well agitated and saturated with oxygen. However, below 0.4 cm/s, reaeration rates were found to be offset by sediment interaction and root respiration, leading to depletion in DO levels. Turbulence induced by wind and wave action and cascading flow, such as over weirs or natural elevation changes, may also be important aspects of surface water aeration in wetlands [5-222, 5-223].

In tidal wetlands, DO replenishment is a function of the tidal flushing rate as well as flow velocity [5-55]. Main flow channels tend to benefit the most. Water within isolated pond areas is not likely to be renewed with oxygen-rich water as frequently. Such areas are more highly suceptible to biological activity and may experience drastic depletion of DO in summer and supersaturation in winter and spring [5-55].

Within wetland systems, DO levels may influence other water quality processes [5-47]. These include:

(1) Denitrification reactions which occur to a significant extent when low DO or anerobic conditions exist;

(2) Changes in the oxidation-reduction potential within sediments leading to the remobilization of heavy metal ions previously bound into stable complexes under aerobic conditions.

o Electrical Conductivity and Salinity

Electrical conductivity (EC) serves as a good indicator of the source of freshwater in wetlands, such as precipitation, runoff or ground-water inflow. Values less than 80 μmho are generally indicative of a perched water table condition, whereas values greater than 80 μmho tend to indicate the presence of continuous ground water [5-216]. In his study of Minnesota peat wetlands, Verry found low EC to be related to increased streamflow, probably due to dilution [5-216].

In the study of Jewell Pond [5-217], EC values of 40 to 70 μmho were observed. Values tended to increase during spring and summer, reach a peak in late July or August and decrease in fall and winter.

The Holland Marsh study [5-217] provided contrast between EC values from cultivated and uncultivated wetlands. Significantly higher EC in both subsurface drainage (90 percent) and surface runoff (88 percent) was observed in the cultivated area. The proximity of snow melt to bare soil in the cultivated area was considered responsible for this difference. The exposure of bare soil provides greater opportunity for the solution of ions than in the naturally vegetated marsh. In addition, greater evaporation rates from the cultivated wetland area were also suspected to be contributing factors.

Salinity refers to solutions with high concentrations of soluble salts, usually predominated by sodium cations and chloride anions. Seawater contains relatively constant concentrations of these major ions and total salinity can be calculated on the basis of sodium or chloride ion concentration. The ionic composition of freshwater is extremely variable and thus cannot appropriately be given on a salinity basis.

Saline environments are usually related to areas with tidal exchange or seawater intrusion. In inland areas, wetlands without outlet or high evaporative stress, can also produce saline environments, which in some cases have higher chloride concentrations than seawater. High salt concentrations can cause osmotic stress to plants, leading to water loss from plant cells, and specific ions may be toxic to some plants. Plants may take up salts to regulate the internal osmotic balance with external conditions. However, the development of adaptation mechanisms, as discussed below is common among many types of wetland vegetation.

In general, sedges of Carex sp. and Cyperus sp. thrive only in freshwater; cattails and rushes can tolerate brackish waters up to about 25 ppt; and reeds, bulrushes, pickleweeds and cordgrasses can tolerate seawater concentrations (33 to 35 ppt) [5-219]. All of the floating and submerged aquatic vegetation species studied are found only in freshwater environments although some will tolerate mildly brackish conditions.

o Color

Wetland environments exhibit large variations in water color due to interactions between sediments, microorganisms and living and decaying plant matter. Water color may range from pale yellow to dark oranges and browns, particularly where bark tannins may contribute strong effects. In most cases, the color of the water is not necessarily indicative of poor water quality. However, in treatment schemes using vegetative systems, it should be noted that effluent waters can contain wide color ranges that may or may not be desirable for other uses, such as drinking water.

Large variations in color were measured in the Wisconsin, Minnesota and New Hampshire studies [5-47, 5-216]. Values given in color units (one color unit is equivalent to the color of 1 mg/l of platinum as K_2PtCl_6) ranged as follows:

- o Wisconsin marsh - 20 to 120 color units;
- o Minnesota peatland - 75 to 890 color units;
- o New Hamphsire ponds - 80 to 150 color units.

Values were found to vary inversely with streamflow in the Minnesota peat wetlands [5-216]. Lee et al., [5-47] also found higher values associated with the breakdown of plant materials at the end of the growing season, when low flow conditions prevailed. The source of color is the decomposing organic matter though only a small group of organics are responsible for imparting visible color [5-47].

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SECTION 6

HYDROLOGIC PROCESSES

Hydrology plays an important role in wetland and upland areas in terms of ecosystem characteristics and pollutant removal capabilities. The primary focus of this study is on wetland systems. The first part of this section is devoted to an examination of hydrologic processes and characteristics of wetlands.

WETLAND HYDROLOGY - CHARACTERISTICS AND PROCESSES

Wetlands occur in a wide range of climatic, topographic, geologic and hydrologic settings, encompassing diverse ecosystem types. Examples include: northern peatlands, bogs, and cattail marshes; pocket marshes and swamplands of the south; salt and freshwater tidal marshes of the Atlantic, Pacific and Gulf Coast regions; and marshes associated with high-energy rivers such as the Mississippi [6-1]. These fall generally into the hydrogeologic categories established by Cowardin et al. [6-2] namely, (a) riverine, (b) lacustrine, (c) palustrine and (d) estuarine (tidal). Even within these categories, diversity of hydrogeologic conditions are exhibited by wetlands.

A clear knowledge of wetland hydrology is crucial to an understanding of the wetland environment and its potential utility in the assimilation of water-borne pollutants. Two aspects should be considered:

(1) Watershed hydrology, i.e., how wetlands relate to the watershed, of which they are an integral feature, in terms of hydrologic response;

(2) Ecosystem hydrology, i.e., how hydrologic factors within wetlands relate to ecosystem characteristics and processes.

Watershed Hydrology

Wetlands are not isolated landmarks but are rather transition zones occurring in response to specific watershed or basin-wide hydrogeologic conditions. They exist primarily in flat to gently sloping terrain, are periodically flooded, and have ground water at or above the surface of the ground for major portions of the year. Beyond this cursory understanding, very little has been done to rigorously assess the relationships between wetlands and watershed hydrology. The study of these relationships is receiving attention due to increased interest in the connection between land development activities, water pollution control, water supply development and wetland values [6-3, 6-4, 6-5].

In connection with land use planning studies in New England, O'Brien and Motts [6-5] sought to establish a system for classifying and evaluating hydrogeologic conditions of wetlands. The objective was to identify key watershed factors which produce significant hydrologic responses in wetlands. This would allow better assessment of impacts of potential land use decisions. The listing of significant factors developed for wetlands typical of the New England area are presented in Table 28. They are arranged under three main categories: geologic, hydrologic and topographic.

A brief review of the various factors cited in Table 28 provides insight into the hydrogeologic characteristics which distinguish Wetland systems and their respective relationship to surface and ground-water regimes.

Geology

Geologic conditions are a major determinant of the structure of the watershed and corresponding runoff characteristics. The nature and extent of geologic formations influence subsurface water movement and exchanges with wetlands. A study by Verry [6-6] on Minnesota peatlands showed that geology has a profound impact on wetlands in terms of (a) hydrologic inputs, and (b) sources and quality of water. The hydrologic connection between wetlands and bedrock can be an indication of the permanence of water supply to the wetland.

Hydrologic Factors

o Hydrologic Position

From a hydrologic standpoint, the occurrence and level of ground water is the most significant factor affecting wetland conditions and hydrologic responses [6-5]. This may be understood in terms of the hydrologic position of the wetland in relationship to the principal zone of saturation. Four typical conditions are depicted in Figure 7:

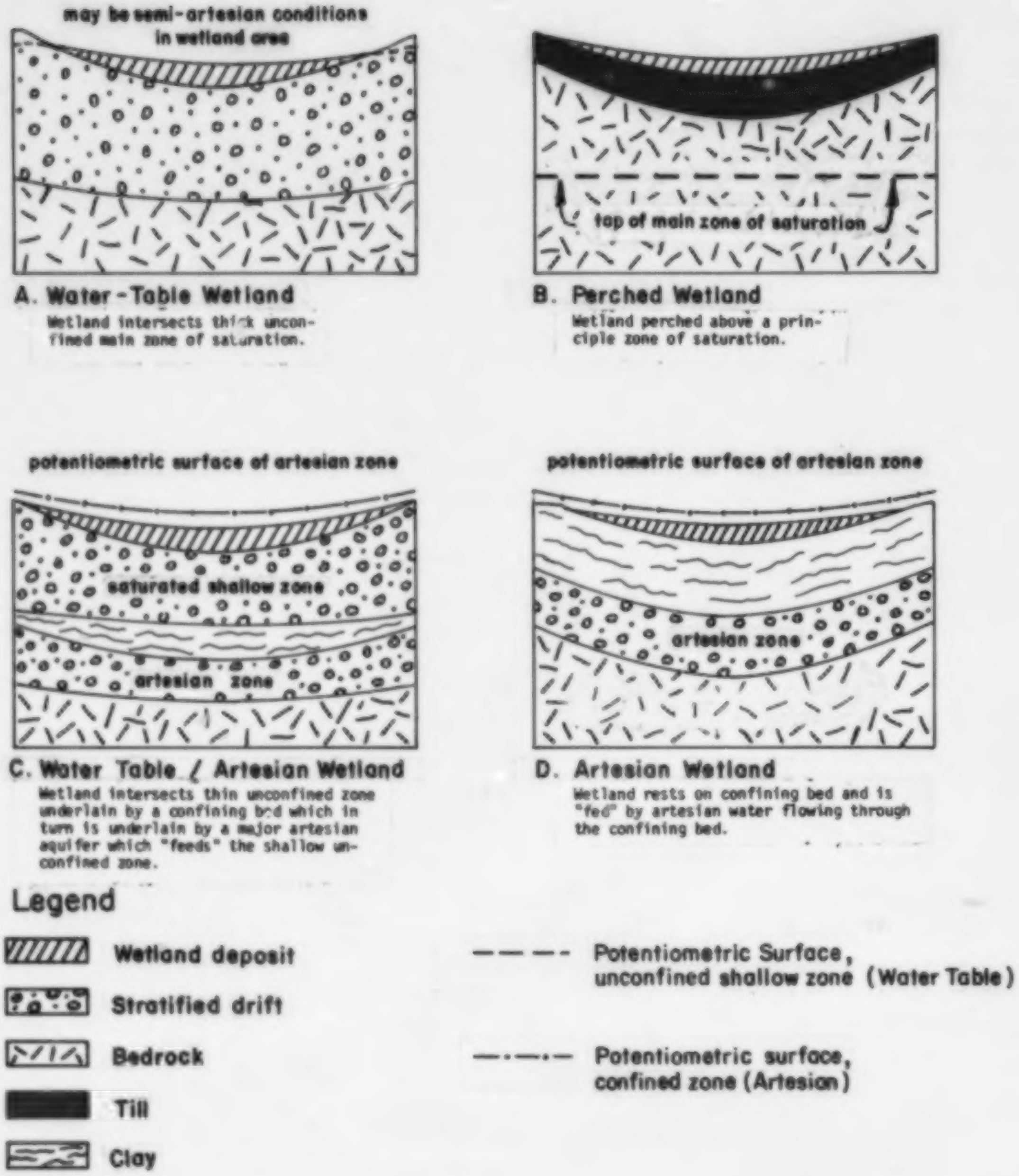
- water table wetland;
- perched wetland;
- water table/artesian wetland;
- artesian wetland.

With respect to hydrologic response, those wetlands perched above the main zone of saturation are in a position to recharge ground water, while those in contact with underlying ground water usually serve as a discharge area.

TABLE 28. SIGNIFICANT WETLAND WATERSHED FACTORS[a]

Factor	Component	Subcomponents
Geologic	Surficial material	Till Stratified draft Lake bed Alluvium
	Thickness of surficial material	Less than 7.5 m Between 7.5 and 15 m Greater than 15 m
	Bedrock	Igneous and metamorphic bedrock Sedimentary bedrock
Hydrologic	Hydrologic position	Perched wetland Water table wetland Water table/artesian wetland Artesian wetland
	Permeability of the organic layer	High permeability (generally associated with peat) Low permeability (generally associated with muck)
	Depth of water in the wetland	Water stands above the surface Water lies below the surface
	Transmissivity of aquifer	Low transmissivity (less than 125 $m^3/(m)(d)$) Moderate transmissivity (125 to 500 $m^3/(m)(d)$) High transmissivity (greater than 500 $m^3/(m)(d)$)
	Ground water outflow	
	Water quality	Water on the wetland surface Water in ponds and streams associated with the wetland Water in the organic deposits Water below the organic deposits (unconfined zone) Water in deep artesian aquifer Water in bedrock
Topographic	Topographic position	Wetland lies near mouth of basin Wetland located on the floodplain Wetland near the basin divide
	Wetland size	

a. Source: reference 6-5.


may be semi-artesian conditions in wetland area
top of main zone of saturation
potentiometric surface of artesian zone
saturated shallow zone
artesian zone
potentiometric surface of artesian zone
artesian zone
Legend
▨ Wetland deposit
Stratified drift
Bedrock
■ Till
Clay
— — — — Potentiometric Surface, unconfined shallow zone (Water Table)
—·—·— Potentiometric surface, confined zone (Artesian)

A. Water-Table Wetland
Wetland intersects thick unconfined main zone of saturation.

B. Perched Wetland
Wetland perched above a principle zone of saturation.

C. Water Table / Artesian Wetland
Wetland intersects thin unconfined zone underlain by a confining bed which in turn is underlain by a major artesian aquifer which "feeds" the shallow unconfined zone.

D. Artesian Wetland
Wetland rests on confining bed and is "fed" by artesian water flowing through the confining bed.

Figure 7. Wetland relationships to the principal zone of saturation (hydrologic position).[a]

a. Source: reference 6-5.

- **Transmissivity and Ground-water Outflow Permeability**

Relatively little study has been made of the relationship of wetland soil permeability to basin-wide hydrologic influences [6-5]. The accumulation of peat results in decreased surface soil permeability and reductions in recharge to ground waters [6-7]. In contrast, cypress domes and other seepage wetlands are generally characterized by high permeability rates. This factor is of major importance with regard to water movement, vegetative responses and filtering capabilities [6-8].

The transmissivity of the underlying aquifer becomes important in assessing the character of sustained subsurface flow from the wetland basin and in assessing possible impacts on water quality and quantity. The ground-water outflow from wetlands is a difficult parameter to gauge, but an important one in constructing the hydrologic budget. It is critical in assessing impacts during low flow conditions and in estimating sustained water yield in wetland areas.

- **Depth of Surface Water**

The depth of surface water in wetlands may be an important factor insofar as it affects hydrologic responses of wetlands to runoff events. Wetlands act generally as regulating reservoirs during high runoff periods, providing storage to retard peak flows. The depth of standing water influences the available storage capacity. Additionally, the depth of water may have a bearing on the manner in which runoff waters circulate. Overland or sheet flow conditions will predominate once normal flow channels are overtopped [6-9].

- **Water Quality**

An expanded discussion of wetland-water quality relationships is provided in Section 7. O'Brien and Motts [6-5] distinguish six hydrologic zones where water quality may be of concern:

- on the wetland surface;
- in ponds and streams associated with wetlands;
- within organic deposits;
- in the unconfined zone below organic deposits;
- in deeper artesian aquifers;
- in bedrock below surficial deposits.

- **Topographic Position**

The ability of a wetland to modify runoff depends upon its position within a basin. The nearer the wetland to the basin outlet, the larger its tributary area and the greater its effect on watershed runoff response. In terms of ground-water relationships, recharge is likely to occur along the watershed perimeter, whereas the ground-water discharge area can be expected at the basin mouth [6-5].

Ecosystem Hydrology

The fact that wetlands are largely a function of prevailing hydrogeologic conditions is clear. Moreover, the reciprocal effects of wetlands on watershed hydrologic responses have been elucidated. Hydrologic factors also play a significant role in determining ecosystem characteristics within wetland areas.

Relatively little investigation has been made into the quantitative aspects of wetlands hydrodynamics. The most thorough examination of the subject is that by Gosselink and Turner [6-1] on qualitative relationships between hydrology and ecosystem traits. They propose a conceptual model, as shown in Figure 8, displaying the connection between wetland hydrology and physical, chemical and biological characteristics of a wetland ecosystem. Gosselink and Turner identified four key hydrologic parameters of most significance to wetland biota:

(1) Source - determines ionic composition, oxygen saturation and pollutant load;

(2) Velocity - affects turbulence and the ability to carry suspended particulate matter;

(3) Renewal rate - describes the frequency of replacement of the water considering water depth, volume, frequency of inundation and velocity;

(4) Timing - constitutes the frequency of inundation (daily, seasonal) and its regularity or predictability.

These primary hydrologic factors affect physical and chemical aspects which in turn influence ecosystem development. The specific ecosystem characteristics identified by Gosselink and Turner [6-1] fall into four categories:

- species composition and richness;
- primary productivity;
- organic deposition and flux;
- nutrient cycling.

Some of the key processes and relationships linking hydrologic factors and ecosystem characteristics are summarized in Table 29. A general conclusion that may be drawn from the information in Table 29 is that water movement through wetlands has a positive influence on the health and character of the ecosystem.

As indicated in Gosselink and Turner's conceptual model (Figure 8), the physical and biological development of a wetland ecosystem may exert certain feedback influences upon the hydrologic regime. These are

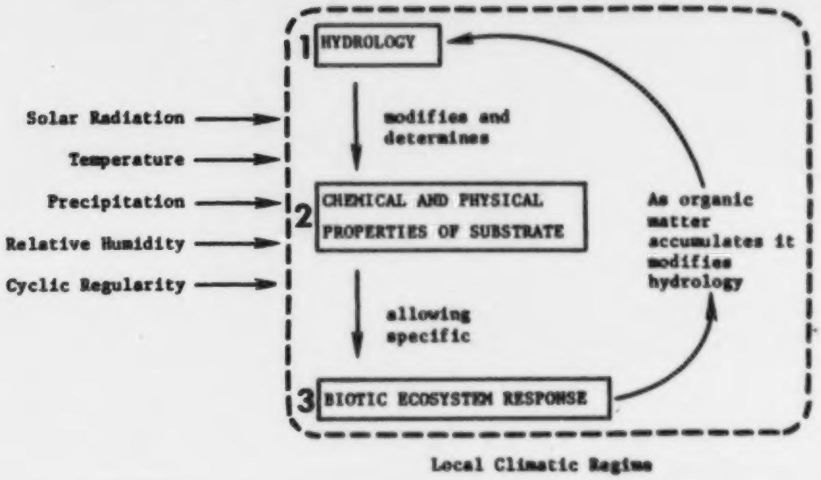

1 HYDROLOGY
modifies and determines
2 CHEMICAL AND PHYSICAL PROPERTIES OF SUBSTRATE
allowing specific
3 BIOTIC ECOSYSTEM RESPONSE
As organic matter accumulates it modifies hydrology
Solar Radiation
Temperature
Precipitation
Relative Humidity
Cyclic Regularity
Local Climatic Regime

Figure 8. General conceptual model of the role of hydrology in wetland ecosystems.[a]

a. Source: reference 6-1.

TABLE 29. WETLAND ECOSYSTEM RESPONSES TO VARIOUS HYDROLOGIC FACTORS

Ecosystem characteristics	Source	Hydrologic factors		
		Velocity	Renewal rate	Timing
Species composition and richness	o Nutrient overload may alter species diversity (6-10)	o Affects distribution & deposition of sediments, influencing elevation and plant zonation (6-11) o Species richness found to increase directly with velocity (6-12)	o Provides vehicle for water movement and circulation (6-1) o Uniform mixing leads to monospecific stands of vegetation(6-1) o Diversity tends to increase with elevation, which is influenced by flooding duration & depth (6-12)	
Primary productivity	o Affects availability of dissolved nutrients for plant growth (6-1) o Sediment inflow increases substrate density, leading to more vigorous plant growth (6-13) o Salts, pesticides and other toxins detrimental to productivity (6-1,6-14)	o Increased velocity related to greater sediment input and increased plant growth(6-1) o "Edge-effect"--stimulation of production along channels due to increased velocity(6-15,6-16,6-17) o Affects flow and availability of toxins (6-18) o Stagnant waters linked to anaerobic conditions and plant stress (6-14,6-19) o Dissolved oxygen related to velocity (6-20)	o Availability of water seems to control lateral spread of ombrotrophic bogs(6-21) o Availability of nutrients for plant growth related to availability of water(6-1) o Regular renewal of water in tidal areas minimizes salt accumulation and plant stress(6-18) o Regular renewal supplies O_2, minimizing stressful anaerobic conditions; depth & duration of flooding most important (6-1, 6-19)	o Timing or seasonality of rain input may affect lateral and vertical spread of ombrotrophic bogs (6-22) o Frequency of flooding influences availability of toxins to wetland flora and fauna(6-1)

TABLE 29. (continued)

Ecosystem characteristics	Source	Hydrologic factors		
		Velocity	Renewal rate	Timing
Organic deposition and flux		o Rate of total particulate and total organic export directly proportional to flow rate (and velocity) (6-23)	o Increased flow rate related to greater silt input and organic matter outflow (6-23)	o Flooding frequency directly related to silt input and organic matter outflow (6-18) o Soil organic concentration increases on gradient from actively flooded stream banks to less actively flooded inland high marshes (6-24)
Nutrient cycling	o Influences nutrient loading and acidity (6-25) Ombrotrophic bogs--nutrient poor Minerotrophic bogs--nutrient rich		o Influences mass loading, transport and flux of nutrients (6-1)	o Nutrient flux related to timing of flooding with respect to plant growth cycle. (6-1)

associated with various aspects of wetland succession. Classical feedback loops involve organic accumulation, sediment deposition and nutrient flux.

The growth of vegetation within wetlands acts to increase the depth of sediment. Plant production supplies organic matter which is available for deposition and incorporation in sediments as peat. The vegetation itself acts as a physical silt trap, increasing the sedimentation rate. With increasing elevation, the frequency and depth of inundation decreases, leading to still greater peat and organic matter deposition. This vertical growth produces two distinct hydrologic responses: (a) blockage or diversion of flow; and (b) decreased percolation of surface waters through organic deposits (peat) [6-12, 6-17].

Also associated with the vertical succession of wetlands is the tendency to reduce the renewal rate and therefore the nutrient flux across marsh boundaries. This results in effectively closing the nutrient cycle [6-1].

The maturing processes described here can and do result in dramatic alterations in biological communities and hydrologic regimes. Often, however, this development is interrupted by periodic inundation which tends to alter and simplify the ecosystem. This has been referred to as "pulse stability" [6-26].

HYDROLOGIC MANAGEMENT OF WETLANDS FOR POLLUTANT REMOVAL

The importance of the hydrologic regime in relation to wetland ecosystem characteristics and processes has been described. Hydrologic factors play an equally significant role in governing wetland functions affecting pollutant removal. A detailed discussion of wetland pollutant removal mechanisms, including the influence of various hydrologic factors, was presented earlier in Section 4. Table 30 provides a summary of the key hydrologic relationships which may influence wetlands strategies for pollutant control.

TABLE 30. RELATIONSHIP OF HYDROLOGIC FACTORS TO POLLUTANT REMOVAL IN WETLANDS

Hydrologic factors	Major pollutant removal mechanisms			
	Sedimentation	Aeration	Biochemical transformations	Soil adsorption
Velocity and flowrate	Settling of particulate matter is primary mode of pollutant removal, largely influenced by flow velocity (6-27,6-28) High runoff flows are threat of washout of nutrients and decomposing organic matter (6-29, 6-31)	Reaeration coefficient varies linearly with velocity (6-34, 6-35,6-20) DO levels a function of tidal flushing and rate of water movement (6-38) High rates of nitrification expected with flowing water (6-43)	Nutrient balance related directly to hydrologic balance (6-41) High N and P levels associated with high flows--indicative of leaching and flushing of decomposing organic matter (6-29)	High flow rates limit contact with soils and seepage potential (6-8)
Water depth and fluctuation	Shallow depths provide for more effective settling (6-30)	Reaeration varies inversely with depth (6-34,6-35) Ponded water contributes to anaerobic conditions, particularly in deeper marshes (6-34,5-44) Fluctuating water level and exposure of soil to atmosphere has positive effect on nitrification; little or no effect on denitrification. (6-43) Water levels responsible for O_2 status of wetlands, and resultant amount of soil microbial activity (6-41) Fresh and saltwater marsh soils capable of reducing NO_3 in ponded floodwaters through denitrification (6-39,6-40)	Water hyacinth exhibit best uptake w/shallow water. (6-32) Many emergent macrophytes perform best at maximum depth . (6-32) Ponded water may encourage volunteer wetland species and biological activity where previously nonexistent. (6-8)	Maximum P immobilization achieved through soil and organic litter contact at lower water levels. 6 cm depth significantly more effective than 30 cm in peatlands. (6-31)
Detention time	Detention and flow equalization minimize downstream erosion. (6-5) Particulate matter deposition directly related to residence time. (6-31)	BOD removal a function of residence time in artificial marshes (6-45) Denitrification in flooded marsh soils a function of detention time. (6-40)	Nutrient removal directly related to detention time. (6-31,6-38,6-45)	
Circulation and flow distribution	Flow distribution most important factor in determining effectiveness of meadowland treatment--sheet flow most desirable. (6-9)	DO replenishment associated with flushing and flow circulation from tidal action and upstream discharges. (6-38)	Well-defined channels allow passage of water and wastewater without significant interaction with aquatic plant communities. (6-29)	Even distribution of wastewater over land surface enhances soil contact and P removal. (6-31) Routing of flow to encourage contact with mineral soils improves P removal. (6-41,6-42)

(continued)

TABLE 30. (continued)

Hydrologic factors	Major pollutant removal mechanisms			
	Sedimentation	Aeration	Biochemical transformations	Soil adsorption
Circulation and flow distribution (Cont'd.)	Beaver dams beneficial in dispersing flow over meadowlands to enhance sedimentation and filtration. (6-9)		Broad flow distribution leads to large effective areas for biological contact and high removal rates. (6-31)	Soil-water interface essential as major site of sorption, deactivation and denitrification. (6-8)
	Circuitous flow paths and sheet flow lead to large effective areas and good sedimentation. (6-9,6-31,6-32)			
Turbulence and wave action		Reaeration a function of channel roughness. (6-34,6-35)		
		Wind can be important in keeping localized parts of wetland water aerated and mixed. (6-36,6-37)		
Seasonal and climatic factors	Most dramatic sediment reductions associated with storm episodes in meadowlands--more even flow distribution over wetland. (6-9)	Pond and pond-like areas nearly depleted of DO in summer and supersaturated in winter and spring. (6-29,6-38)	Freezing promotes release of N and P from plants and soils for subsequent washout. (6-28,6-29)	
		Nitrifying and denitrifying bacteria limited in activity during drought or low water temperatures. (6-28)	Spring snow melt appears to have flushing effect on nutrients. (6-28,6-29)	
			Best nutrient removal in spring-summer growth period. (6-32)	
			First flush of wet season may wash out decaying organic matter from prior winter. (6-29)	
			Temperate systems experience winter release of nutrients. (6-46)	
			Seasonal rainfall and runoff important as diluting agent and to encourage water circulation. (6-47)	
Soil saturation		Lowering of water table leads to aeration, mineralization and greater mobilization of N. (6-29,6-48)	Level of soil saturation controls microbial activity. (6-41)	

(continued)

TABLE 30. (continued)

Hydrologic factors	Major pollutant removal mechanisms			
	Sedimentation	Aeration	Biochemical transformations	Soil adsorption
Soil saturation (Cont'd.)		G.W. table within 5-10 cm of wetland surface promotes anaerobic (reducing) conditions. (6-49)	Higher respiratory activity and biological decomposition with well drained soil conditions. (6-41)	
		Lowering of water table allows aeration and decomposition of organic matter and slug release of nutrients in next flush of runoff. (6-29)		
Permeability and ground water movement			Source and amount of g.w. inflow influences pH and resultant solubility and precipitation reactions. (6-6, 6-29)	Seepage wetlands may act similarly to flood-irrigation forage grass systems in virtual completeness of P removal. (6-8)
				Nitrogen removal through nitrification/denitrification may be excellent in surface soils and shallow seepage zone. (6-8,6-27)

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SECTION 7

CASE STUDY REVIEW

Growing interest in the natural resource value of aquatic and terrestrial ecosystems has spawned considerable research activity and numerous symposia over the past 10 to 20 years. It has arisen out of (a) concern over the susceptibility of such systems to land development and pollution sources, and (b) scientific curiosity as to the role of natural biological systems in assimilating various types of wastewater.

Literature related to wetland vegetative systems covers a wide range of subjects. This includes (a) fundamental research regarding function, biochemical processes, productivity, nutrient cycling and physical relationships, as well as (b) a growing number of site-specific investigations of pollution impacts and treatment capabilities. The focus of this review is on the latter. Preceeding sections of this report reference particular research efforts dealing with fundamental processes and characteristics of wetland and upland vegetative systems.

The following review is intended to provide an understanding of how different types of vegetative ecosystems in different geographical regions have been observed to influence water quality and provide treatment of point and nonpoint source wastewater flows. Processes, relationships, function and operation of particular systems are treated in greater depth in the analysis of vegetative and hydrologic practices in Sections 5 and 6.

USE OF WETLANDS FOR WASTEWATER TREATMENT

Considerable attention has been devoted, recently, toward the use of wetlands for wastewater treatment and nutrient assimilation, particularly in North America and Europe [7-1, 7-2]. In Europe, research and applications involving wetlands and vegetative waste treatment date back to the early work of Kathe Seidel and others at the Max Planck Institute, in the early 1950's [7-3]. In the U.S., the past ten years have seen promising results with experimental applications of wastewater to peatlands in Michigan and Wisconsin [7-4, 7-5, 7-6, 7-7], tidal marshes in Louisiana and New Jersey [7-8, 7-9], cattail marshes in Wisconsin [7-10, 7-11, 7-12], cypress domes in Florida [7-13, 7-14, 7-15], and other natural and artificially constructed wetland systems in various regions of the country [7-16, 7-17, 7-18, 7-19, 7-20]. A discussion of the experience and findings of several major investigations with different types of wetlands are presented below. A summary of observed pollutant removal efficiencies for most of the referenced studies is provided in Table 31 following the narrative discussion.

Northern Freshwater Marshes

o Brillion, Wisconsin

Much work has been carried out in Wisconsin concerning the feasibility of using natural and artificial marshes to provide secondary and tertiary treatment of wastewaters. Investigations by Fetter et al., [7-10] on a natural cattail (Typha) marsh at the city of Brillion provide some of the most informative practical experience with freshwater marshes subjected to long-standing wastewater discharges. The 15-month study showed significant improvement in wastewater and stream quality after passage through the marsh. In terms of critical nutrients, a 51 percent reduction in concentration of nitrate and a 13 percent reduction for phosphorus were observed. Additionally, annual net retention of phosphorus, due largely to precipitation into organic sediments, was estimated to be perhaps as much as one-third of the amount entering the marsh. Removal of oxygen-demanding substances and suspended solids also showed very positive results. Despite the apparent success with natural systems, the researchers favor efforts to construct artificial marshes rather than discharging wastewaters to natural wetlands.

o Hay River, Northwest Territories, Canada

An investigation similar to the Wisconsin study was conducted in the Northwest Territories of Canada at a swampland vegetated primarily with sedges (Carex) [7-21]. The study area had been receiving discharges of secondary effluent from the town of Hay River (population 3,000) for many years. Substantial removal of wastewater pollutants was found to occur in the swamp, to the extent that after passage through 60 percent of the wetlands (3,600 m of channel), water quality was "nearly indistinguishable" chemically and biologically from that of unaffected waters in the vicinity. Pollutant removal efficiencies were generally in excess of 90 percent during the summer flow period. Whether nutrient removal is limited by rate of supply from the sewage, or by the rate of uptake by macrophytes or sediments was not determined. Of ecological significance, however, was evidence of reduced biological diversity over 40 percent of the wetland. Water quality was also measureably altered within the swamp as a result of wastewater input to this area.

o Cootes Paradise, Ontario, Canada

In Ontario, Canada, a tidally affected lacustrine wetland was investigated by Murdoch and Capobianco [7-22] to determine the effects of long-term wastewater discharge. The study area (Cootes Paradise) consisted of a 5.2 sq km wildlife sanctuary of which 1.7 sq km was open water and the remainder, marsh areas and woodland. Marsh vegetation included many emergent and submerged plants, the major species being Glyceria grandis, Myriophyllum, Ultricularia, Ceratophyllum and Lemna. In comparison with prior vegetative surveys, the area directly downstream of the wastewater discharge showed an apparent decrease in biological diversity with the loss of a stand of Typha sp. A pronounced build-up of certain heavy metals, nutrients and organic carbon was also found in the marsh soils of this area. Correspondingly higher levels of heavy metals were also observed in the aquatic plants of this impacted section of the wetland. The need to direct attention to the possibility of heavy metal transfer to waterfowl and other parts of the food chain was suggested by the investigators.

o Great Meadows National Wildlife Refuge, Massachusetts

In conjunction with feasibility studies of the overall concept of wetland disposal of wastewater in Massachusetts, Yonika and Lowry [7-23] investigated the effects of an existing wastewater discharge to the Great Meadows National Wildlife Refuge in Concord. This freshwater marsh wetland has been the recipient of sewage discharges since the early 1900's. Concerns recently expressed about odor problems and organic impacts on the wetland prompted the investigators to select Great Meadows as a study site. Field investigations and evaluations were conducted with respect to wetland hydrology, nutrient dynamics, and water quality. Analyses were also conducted on sediment quality, invertebrate populations and vegetative community. Results of the testing program revealed:

(1) Great difficulty in attempts to model hydrology;

(2) Different renovating capabilities in different sections of the marsh and, more significantly, in different seasons of the year;

(3) Significant nutrient retention in the sediment;

(4) Dense luxuriant plant growth within a 60 to 75 foot radius of the wastewater effluent pipe and evidence of a "thermal plume" in winter;

(5) Predominance of pollution-tolerant tubificid worms within 250 feet of the discharge point;

(6) Conditions at the marsh outlet to the Concord River similar to recovery zone downstream of a sewage discharge.

Southern Freshwater Marshes

Several investigators have examined the usefulness and impacts of wastewater disposal in freshwater marshes in the southeastern portion of the U.S. The studies have met with variable results.

- **South Florida Everglades**

In an investigation of sawgrass wetlands in the South Florida Everglades, Steward and Ornes [7-24], found that experimentally applied levels of wastewater overloaded the marsh ecosystem with phosphorus in a very short time (8 weeks). The low nutrient requirement of sawgrass was felt to be a significant factor limiting the assimilative capacity in this particular wetland situation. Changes in the composition of the plant community indicated that undesirable effects on the unique structure of the Everglades can result from the increased supply of nutrients associated with wastewater discharges.

- **Clermont, Florida**

Other freshwater swamps and marsh areas were investigated in Florida with more encouraging results. In 1977-78, a 10-month pilot study was conducted for the city of Clermont, Florida, to determine the effect of varied loadings of secondary wastewater effluent on the productivity, and nitrogen and phosphorus budget of a freshwater marsh [7-17]. The study site was within a 32-ha marsh, composed primarily of emergent aquatic macrophytes. Arrowhead, pickerel weed, panic grass and marsh hibiscus were the dominant species. The study successfully demonstrated the viability of utilizing the marsh as a tertiary system for upgrading wastewater effluent quality. Specifically, the initial year of operation showed major reductions in nitrogen and phosphorus, to concentrations comparable with background levels. The apparent reductions in phosphorus were due to accumulation and storage in the soil complex, roots and dead standing matter. Studies of soils and vegetation showed a possible higher rate of peat production and greater plant uptake of phosphorus and nitrogen in areas receiving wastewater discharges.

- **Wildwood, Florida**

In a University of Florida research study [7-16], a hardwood swamp which had been receiving wastewater effluent and surface runoff from the City of Wildwood (population 2,500) for about 20 years was investigated. Three contiguous wetland areas totaling about 200-ha were examined. The wetland sequence consisted of a marsh community vegetated with duckweed, cattail and willow, followed by two hardwood swamps. Water quality studies showed reduction of nutrients through the wetlands to levels equal to or less than those in the receiving waters of Lake Panosoffkee and in an adjacent control swamp. High levels of nutrient removal were associated with increased productivity of vegetation in the swamp. Statistically significant growth rate increases were noted for cypress and ash. Unlike other wetland studies [7-25], nutrient buildup in sediments was not found. The inflow of urban runoff was suspected of causing disturbance of sediments in some areas of the wetland, resulting in inconsistent ammonia nitrogen readings. Relatively low concentrations of heavy metals in marsh and swamp waters reflected the low levels discharged from the sewage treatment facility.

- **Cypress Wetlands, Florida**

Cypress wetlands, commonly referred to as "domes", have been the subject of considerable interest and investigation for wastewater disposal feasibility [7-15]. The Center for Wetlands at the University of Florida has been testing several cypress domes near Gainesville. The results indicate rapid rate of nutrient uptake and little change in the nutrient quality of underlying and downstream ground waters as a result of wastewater application. Seasonal variations in rainfall make it appropriate to adjust wastewater loading rates on a seasonal basis in these wetlands. A comparison of vegetation in the cypress domes showed:

(1) Extensive growth of duckweed in the cypress domes receiving wastewater;

(2) Varied aquatic plants in domes situated down-stream of the domes receiving wastewater;

(3) Poor aquatic productivity in the natural domes unaffected by wastewater application operations.

Northern Peatlands

The potential for treating wastewater flows in natural peatlands in midwestern and northern portions of the U.S. has attracted the attention of engineers and researchers. The most extensive work was that performed by the University of Michigan Wetlands Ecosystem Research Groups. The main focus of the research has been at Houghton Lake, Michigan.

- **Houghton Lake, Michigan**

Study of the Houghton Lake peatland began as a pilot investigation to determine the feasibility of utilizing the wetland for final tertiary treatment and discharge of wastewater effluent from Lake, Denton and Roscommon Townships [7-4, 7-5]. The Tri-Township was in the process of developing regional wastewater facility plans in the early 1970's when the idea to make use of the wetland was first considered. Between 1972 and 1977, Kadlec and others conducted investigations of a 1700-ha wetland site southwest of Houghton Lake, applying up to 380 m^3/d of pond stabilized wastewater. The study site was comprised mainly of sedges (*Carex* sp.), with scattered shrubs (*Salix* sp.) and herbs (*Aster* sp.). Field studies included analysis of surface water quality, plant productivity and nutrient status of plants and soils. Results indicated that peatlands have potential as biological filters for removal of certain nutrients. This was attributed to several factors:

(1) High nutrient adsorption capacity of organic litter and peat soils;

(2) Slow subsurface movement of water;

(3) High denitrification rates within the water-logged soils;

(4) Nutrient uptake by some plants.

In general, a high degree of contact between surface water, peat and the litter layer has been found to be particularly important to successful wastewater treatment through peat wetlands.

At Houghton Lake, no adverse effects within the wetland as a result of addition of nutrients and other wastewater constitutents were observed. In fact, the added nutrients are considered to be beneficial to an otherwise nutrient-poor wetland [7-20]. As a result of these promising pilot studies, approval was granted and construction began in 1977 on one of the first full-scale operational wastewater facilities using the concept of wetlands for final effluent polishing. The system is currently operating as anticipated, with nutrient reductions of 90 percent within 30 meters of the discharge point.

- **Bellaire, Michigan**

Kadlec and Tilton [7-4, 7-6] have also monitored and evaluated the discharge of wastewater effluent to a 18-ha, partially diked wetland in Bellaire, Michigan. The wastewater is from the village of Bellaire (population 1,000) which produces an annual volume of approximately 114,000 m^3 of sewage. The objective of discharging to the wetland is to protect downstream receiving waters (Lake Bellaire) from excessive nutrient additions. The sewage receives secondary (lagoon) treatment prior to discharge. Releases to the wetland occur from May to November, with holding ponds providing overwinter storage of effluent. The wetland is distinguished from the Houghton Lake facility by the closed canopy of coniferous trees at the Bellaire site. Despite this vegetational dissimilarity, nutrient removals (80 to 90 percent) were found to be as efficient as at Houghton Lake.

Tidal Wetlands

In the late 1960's, Grant and Patrick [7-26] undertook a field investigation of the Tinicum Marsh in Pennsylvania to assess the health and function of the tidal wetland ecosystem in relation to existing wastewater discharges. Three sewage treatment plants discharging into the marsh were known to be contributing to high organic loadings to the wetland. The main focus of the investigation was on nutrient assimilation, reoxygenation of surface waters and wetland productivity.

The chemical analyses and biological field studies confirmed damage to the marsh from the wastewater loadings. In particular, severe bacterial contamination, low oxygen and high nutrient levels were observed. Decreased species diversity in the Tinicum Marsh in comparison to a nearby undisturbed marsh provided evidence of biological damage.

Because of tidal flow and varying volumes of sewage discharges, the researchers found it difficult to assess the distribution of organic loads and the exact pollutant reduction efficiency of the wetland. It was observed, however, that significant reductions in BOD, phosphate, ammonium and nitrate occurred as a result of the tidal excursion of the river water over the marshland, in the time interval of about two to five hours. Although field data showed inconsistencies, probably due to the irregular flow pattern and pollution load mentioned previously, it was evident from the study that these particular marshlands make a significant water quality contribution through reduction of nutrient loads and increases in oxygen content.

Artifical Wetlands

o Germany and Holland

The creation of artificial wetlands for treatment of wastewaters has been investigated for both small and large scale applications in Europe and the U.S. with different types of vegetation and substrate. Some of the most extensive information has been compiled by Kathe Seidel and others at the Max Planck Institute in Germany [7-3]. Here, the functions and role of higher aquatic plants in wastewater purification have been investigated. The work has resulted in the development of a patented wastewater process, the Max Planck Institute (MPI) system, which relies on cultures of selected plant species. The system consists of two parts:

(1) Sludge retention and decomposition, and reduction of certain bacteria counts, by beds of Phragmites plants;

(2) Extensive elimination of dissolved organic and inorganic substances and of bacteria by means of a cascade plantation consisting of Schoenoplectus lacustus or other appropriate plants.

The system has potential for treating small wasteflows, such as those generated at campgrounds, schools, farms, hotels, etc. To date, it has been successfully employed at several locations in the Netherlands. Joost De Jong [7-27] reports on the application of an artificial rush pond at a seasonal campground in Holland. Several geometrical layouts were examined, prior to choosing long ditches that facilitated mechanical maintenance. The system demonstrated very good removal efficiency for BOD, nitrogen, phosphorus and bacteria, using raw sewage influent.

The main pollutant removal functions were related to infiltration-percolation, sedimentation, and plant uptake. Removal of all pollutants was found to vary with detention time, with best results occurring after a minimum of 10 days retention. Decreases in nutrient removal toward the end of the season (August) were attributed to an overloading of nitrogen and phosphorus by this point.

o Long Island, New York

In the U.S., artificial marsh systems were constructed at the Brookhaven National Laboratory in Long Island, New York [7-18, 7-28, 7-29, 7-30, 7-31]. Two closed systems have been under study since 1973: (a) a marsh/pond system and (b) a meadow/marsh/pond system. The component parts were as follows:

(1) Meadows were backfilled with 15 to 20 cm of sand loam and planted with reed canarygrass; these were eventually supplemented with volunteer grasses and weeds;

(2) Marshes were backfilled with 10 to 15 cm of mulch and planted principally with cattails;

(3) An 80 m^2 pond served to stabilize effluent from the marshes and meadows. It supported a healthy growth of duckweed.

Both systems have demonstrated the ability to renovate applied sewage and septage mixtures of up to 80 m^3/d. Measurement to date of all key water quality parameters indicates that both systems produce water suitable for ground-water recharge, over or through vegetated soil, without posing public health hazards. The investigators are strongly in favor of using artificial marsh systems rather than natural wetlands for wastewater treatment purposes [7-28].

The creation of wetlands for wastewater treatment purposes is of growing interest because of the multiple benefits that may be realized. Two particular systems currently in operation illustrate this potential and are discussed below.

o Martinez, California

The loss, over the years, of brackish and fresh water marshes and wetlands in the San Francisco Bay Area, to urban and agricultural development has recently given rise to public interest in the preservation and restoration of marsh lands. The objective has been realized in the bayside city of Martinez, where secondary-treated effluent has been used to create a freshwater marsh [7-19, 7-32]. In meeting waste discharge requirements and wetland objectives, the project has gained substantial public support. The project has provided the following:

(1) A balanced and healthy wetland ecosystem created through the reuse of treated wastewater;

(2) Newly established wetland habitat for numerous species of animals, birds and aquatic invertebrates;

(3) Consistent nitrate removal and seasonal BOD and suspended solids removal;

(4) Increasing educational and recreational usage;

(5) The impetus for the development of similar pilot-scale projects in the San Francisco Bay Area and other areas in California [7-33, 7-34, 7-35].

- **Vermontville, Michigan**

In Vermontville, Michigan, a seepage wetland system was constructed to provide final treatment (phosphorus removal) for pond-stabilized wastewater [7-20, 7-36, 7-37]. The system is operated in a manner similar to flood irrigation, with treatment provided by nutrient uptake in upland grasses, and soil-water interaction as wastewater seeps downward and laterally from the irrigation fields. The grasses are harvested periodically. The irrigated land has become a marsh area, inhabited by a wide variety of volunteer wetland plant species, including cattail, willow, duckweed, grass and goldenrod. During the irrigation season, areas of standing water support a variety of waterfowl and other wildlife. The primary treatment objective of phosphorus removal is obtained for most of the year. The exception is in the late spring and summer irrigation season when the wetland actually increases phosphorus amounts above those of the applied wastewater. This is in part due to inputs of rainfall which cause wastewater to pass over the surface of the marsh, bypassing seepage routes. These results are in agreement with other wetland studies which suggest that the flow-through process without substantial contact and filtration by native soils might prove unsatisfactory for removal of phosphorus.

Aquatic Plant Systems - Water Hyacinth

The use of water hyacinth and other free floating plants for wastewater treatment is generally not considered a wetland system, but rather an aquatic plant system. Much work on these aquaculture treatment systems has been done recently in the southern areas of the U.S. The reader is referred to several case studies [7-38, 7-39, 7-40] presenting information and a general analysis of wastewater treatment by means of water hyacinth and duckweed. Some of the most extensive work on the subject has been reported by Dinges [7-41, 7-42, 7-43] for the treatment at the Williamson Creek Facility in Austin, Texas, and by Wolverton et al., [7-44, 7-45, 7-46] for experimental work at the National Space Technology Laboratories in Bay St. Louis, Mississippi.

The research found that water hyacinth systems are capable of removing high levels of BOD, suspended solids, metals and nitrogen, as well as significant uptake of refractory trace organics [7-38]. Phosphorus removal is limited to plant needs and normally will attain 50 to 70 percent of the influent amount only when careful management and regular biomass harvesting is carried out. In addition to direct plant uptake,

TABLE 31. OBSERVED POLLUTANT REMOVAL EFFECTIVENESS OF WETLAND-WASTEWATER TREATMENT SYSTEMS

System/location	Wetland type	Hydraulic/ hydrologic factors	Application concentration	Removal efficiency	Comments and reference
Brillon, Wisconsin	Cattail marsh	156 ha marsh	NO_3-N 1.17 mg/l	51.3%	o Decrease in NH_3-N and NO_3-N partially attributable to plant uptake.
(Existing discharges)	(deep water)	49.7 km^2 watershed	Total P 3.13 mg/l	13.4% annual	
		Population contributing wastewater - 2,588	BOD_5 26.9 mg/l	80.1%	o P shows a strong seasonal fluctuation. Nutrients mobilized during periods of low biological activity (fall-winter).
		Hyd. loading - 2.2x10^4 m^3 to 4.6x10^4 m^3 per month	SS 154 mg/l	29.1%	
		Ave. water depth of .5 m			o BOD values consistently reduced to below background levels except in winter, when marsh iced over. Anaerobic conditions left organic material undigested.
		Hyd. residence time of 48 to 145 days			
					o Suspended solids inputs were inorganic material from upstream erosion and organic matter from wastewater discharge. (7-10)
Hay River, Northwest Territories (Canada)	Northern swampland	32 ha wetland	NH_4-N 10.3 mg/l	96.2%	o Reductions computed following passage through 32 ha of swampland. Biological diversity and water quality reduced in initial impact area.
		population contributing wastewater - 3,000	Total P 11.0 mg/l	97%	
(Existing discharge)		160,000 m^2 initial impact area for discharge	PO_4-P 9.95 mg/l	97.5%	
			BOD_5 44 mg/l Annual load - 113 gm/m^2	97.7%	o Tests of sediments, roots and stand crop indicate phosphate uptake by dead organic matter in root masses of floating swamp vegetation and also by active plant uptake.
					o Reduction due primarily to photosynthesis by large areas of aquatic moss. Loading of 141 kg BOD/day. (7-21)
Cootes Paradise, Ontario, Canada	Lacustrine marsh	5.2 km^2 wetland	Total N 22.7 mg/l	41%	o Total N reduction related primarily to settling of particulate matter from effluent and some amount of biological uptake.
(Existing discharge)	(deep water)	1.7 km^2 open water			
		0.5 m mean depth of open water			

(continued)

TABLE 31. (continued)

System/location	Wetland type	Hydraulic/ hydrologic factors	Application concentration	Removal efficiency	Comments and reference
Cootes Paradise, Ontario, Canada (continued)		wastewater 30-40% of streamflow	Total P 4.73 mg/l	33%	o Total P reduction related primarily to settling of particulate matter and biological uptake.
		detention time variable, from 2.16 days (spring) to 11.8 days (summer)	Data not available.	Significant accumulations of Cr, Cu, Ni, Pb and Zn above control levels in marsh soils.	o Metal uptake also observed in wetland plants. Main supply to plants judged to be through soils rather than water. (7-22)
South Florida Everglades (Pilot study)	Freshwater marsh (sawgrass)	Equivalent loading of 2.5 cm/wk/ha	Total N 0,11,56 and 112 kg/ha (one-time closing)	Max. plant uptake of 8% after 125 days.	o Plants found to be supplied adequately by available sources of N.
			Total P 10 mg/l (2.5 kg/(ha)(wk))	Dissolved P concentrations remained elevated, 20 times greater than background	o Assimilative capacity of wetland vegetation stressed after 3 weeks, and exceeded after 8 weeks. (7-24)
City of Clermont, Florida (Pilot study)	Freshwater marsh	Four 200 m^2 plots	Total N 10½ month ave. of 1.14 kg/(ha)(d)	97.29% $(NO_3 + NO_2)$	o Dead standing crop and below ground biomass acted as N sinks.
		Loading rates of 1.3, 3.8 and 10.2 cm/wk	Total P 10½ month ave. 1.19 kg/(ha)(d)	97.5%	o Storage in roots, litter and soil complex accounted for P reduction. (7-17)
City of Wildwood, Florida (Existing discharge)	Hardwood swamp	202 ha swamp area	Total N 15.3 mg/l	89.5%	o High levels of N-removal during August and September attributable to plant utilization.
		672 ha watershed	Total P 6.4 mg/l	98.1 - seasonal 13% annual	o P release during dormant period responsible for lower annual removal efficiency.
		570 m^3/d wastewater input	Wastewater Cu - 0.02 mg/l Fe - 0.17 mg/l Mg - 4.1 mg/l Pb - 0.03 mg/l Zn - 0.048 mg/l Urban runoff Cu - 0.01 mg/l Fe - 0.19 mg/l Mg - 3.2 mg/l Pb - 0.02 mg/l Zn - 0.015 mg/l	(Ave. reduction for combined wastewater and urban runoff flow) Cu - no change Fe - 85% Mg - net increase Pb - 60% Zn - 75%	o All influent and control samples relatively low in metal content. No study of uptake or removal mechanisms. No discernible impacts on wetland. (7-16)

(continued)

TABLE 31. (continued)

System/location	Wetland type	Hydraulic/ hydrologic factors	Application concentration	Removal efficiency	Comments and reference
Houghton Lake, Michigan (Pilot studies for new full-scale system)	Peatland	1700 ha wetland 3.9 cm/wk to 6.5 ha site (hyd. loading)	NH_4-N 10.8 mg/l	71%	o Nutrient immobilization due to denitrification and sorption by peat.
			(NO_2+NO_3)-N .36 mg/l	99%	
			Total P .41 mg/l	95%	o P reductions varied with water depth (6-30 cm) and seasonally. Greatest release during late winter and early spring. (7-4, 7-5)
Bellaire, Michigan (New Discharge)	Freshwater marsh (with forest canopy)	20.45 ha wetland May-Nov. loading of 118,740-134,807 m^3	(NH_4+NO_3)-N 4.95 mg/l	91%	o Monthly removal rates varied in response to hydrologic conditions, plant growth, and temperature.
			Total P 3.48 mg/l	97%	o Water samples taken during late winter and early spring indicate a release of P equal to 1.02% of the total added over the period March-November. (7-4, 7-6)
Great Meadows National Wildlife Refuge (Existing Discharge)	Freshwater marsh (deep water)	hydraulic loadings - .009-.024 m^3/s 115 ha watershed 19 ha wetland	yearly average	annual average	o Extreme seasonal variations noted. Highest removal in spring and early summer growing season. Denitrification most significant factor. Low removal rates and release during fall-midwinter (TKN) and winter-early spring (NO_3-N).
			TKN 10.0 mg/l	35%	
			NH_4-N 8.0 mg/l	58%	
			NO_3 3.1 mg/l	20%	
			Total P 2.1 mg/l	47%	o Significant seasonal fluctuations in P removal from 85.7% in early spring to net increase of 72.8% in mid-winter. Variations judged to be related to sediment-vegetation interactions and marsh hydrology.
			Ortho P 1.6 mg/l	49%	o Seasonal variations evident with best removal during June-September. Uptake rates of 6 lbs/(acre)(day).

(continued)

TABLE 31. (continued)

System/location	Wetland type	Hydraulic/ hydrologic factors	Application concentration		Removal efficiency	Comments and reference
Great Meadows (continued)			BOD	38.2 mg/l	67%	o Considerable and inconsistent variations observed. Few correlations found between SS changes and seasons, hydraulic characteristics or biological influences.
			SS - data not available		overall decrease	
						(7-23)
Brookhaven National Laboratory, New York	Freshwater marsh-meadow pond systems	sewage loading of 80 m^3/d	Total N	25.2 mg/l	79%	o Reductions are for the total system (meadow/marsh/pond). The removal abilities of the individual components have not been analyzed.
		60-80% recycled within system	NH_3-N	8.4 mg/l	86%	
(Artificial system)			Kjeldahl-N	19.7 mg/l	81%	
		.2 and .4 ha systems	(NO_2+NO_3)-N	5.5 mg/l	73%	o Meadow-marsh-pond system somewhat more effective than marsh-pond system.
		pond - 1.5 m deep with 946 m^3 capacity	Total P	7.2 mg/l	77%	
		meadows - 3% slope	PO_4-P	4.8 mg/l	77%	
		marshes .5-1m depth	Cr	.05 mg/l	60%	
			Cu	0.7 mg/l	94%	
			Fe	3.6 mg/l	58%	
			Mg	4.3 mg/l	23%	
			Zn	1.3 mg/l	85%	
			BOD_5	170 mg/l	88-92%	(7-30, 7-31)
			SS	353 mg/l	91.5	
Martinez, California	Artificial marsh	6.1 ha wetlands plot			Averages (1975)	o Marsh purposely managed to enhance wildlife rather than to optimize pollutant uptake.
(New full-scale system)	(deep water)	63% open water	NH_3-N	7.0 mg/l	24%	
		37% emergent vegetation	NO_3-N	6.1 mg/l	56%	
		4687 m^3 capacity	NO_2-N	0.38 mg/l	Overall increase	
		4.8 days retention time at 6056 m^3/d	Total organic N	4.1 mg/l	12%	
		current retention time about 10 days				
		current loading is 2839 m^3/d				

(continued)

TABLE 31. (continued)

System/location	Wetland type	Hydraulic/ hydrologic factors	Application concentration	Removal efficiency	Comments and reference
Martinez, California (continued)			PO_4-P 10.4 mg/l	13%	o Best phosphate removal during summer months.
			BOD 19 mg/l	36.8%	o High BOD readings in summer months due to algae.
			SS 22 mg/l	not available	o Measurably lower values during cooler months but significant increase during summer algal blooms. (7-19, 7-32)
Vermontville, Michigan	Artificial seepage wetland	4.6 ha diked irrigation fields	Data on N not available	60% of NO_3	o Reduction occurs through denitrification in the shallow wetland soil.
(new system)		94,661 m^3 of effluent, June-October	Total P 1.0 mg/l (wastewater)	97%	
			Total P 2.1 mg/l (combined wastewater and wetland waters)		o P removal occurs through soil adsorption of wastewater prior to encountering ground water. 95% occurs in upper 3 feet of wetland soil.
					(7-20, 7-36, 7-37)
Williamson Creek Treatment Plant, Texas	Hyacinth pond	0.0585 ha pond	Total N 4.3 mg/l	72%	o Significant nitrogen uptake obtained in both summer and winter.
(Pilot system)		1.0 m depth	NH_3-N 2.1 mg/l	71%	
		1,860 m^3/ha hyd. loading	PO_4-P 15.4 mg/l	31% annual	o Removal of P restricted to plant growth season.
		5.3 days residence time	Data not available on Heavy metals	Significant plant accumulation of Cu, Cr, Fe, Mg, Mn, Ni, Pb, Zn	o Uptake directly from influent and ponded water
			BOD 19 mg/l	81.7%	o Accomplished primarily through reduction of suspended particulate matter capable of producing effluent BOD of less than 10 mg/l.
			SS 46 mg/l	84.5%	(7-40, 7-41, 7-42, 7-43)
National Space Technology Laboratories	Hyacinth pond	2 ha pond	Total N 12.0 mg/l	71.7%	o Regular harvesting of hyacinth needed to maintain effective nutrient removal.
(Experimental system)		1.22 m depth	Total P 3.7 mg/l	57%	
		240 m^3/ha hydraulic loading	BOD 110 mg/l	94.7	o Loaded at 26 kg BOD (ha)(day) without development of odor problems.
		54 days residence time	SS 97 mg/l	89.7%	(7-38, 7-44)

factors which contribute to the wastewater renovating capabilities are (a) an active mass of organisms supported by the root system, and (b) shading of the water surface by plant leaves, limiting light penetration and algae growth [7-38].

LAND TREATMENT OF WASTEWATER

Land treatment of municipal wastewaters has been one of the most extensively studied aspects of sanitary engineering during the past 10 years. The capacity of the soil-vegetation complex for wastewater renovation has long been recognized. The current incentive to continue investigations and expand the usage of land disposal practices can be attributed, in part, to the enactment of the Federal Water Pollution Control Act Amendments of 1972, which clearly focus on reduction of surface water discharge of pollutants, and encourage consideration of land treatment as an alternative treatment method in wastewater facilities plans.

A vast amount of literature is available on various aspects of planning and feasibility investigations, treatment mechanisms and performance, and design criteria and procedures. The most thorough treatment of the subject is provided in the "Process Design Manual for Land Treatment of Municipal Wastewater" [7-48] sponsored by the USEPA, US Corps. of Eng. and USDA in 1977. An exhaustive review of practice and literature pertaining to land disposal methods is provided in that document. In addition, an update of this manual is currently under preparation.

WETLANDS TREATMENT OF SURFACE RUNOFF

Field studies and research concerning the use of wetlands for treatment of surface runoff flows have been extremely limited. The recent interest that has developed is, primarily, a result of the search for innovative methods for controlling nonpoint surface runoff pollutants.

Case Studies

Several major investigations which provide information on the effectiveness of wetlands as stormwater treatment systems are summarized here. They represent a wide diversity of wetlands types and geographical regions. Included are northern peatlands, cypress wetlands, a brackish marsh, high altitude meadows and wetlands detention basins. Brief descriptions of the various wetland-stormwater systems which were the subject of these investigations are provided below.

- **Wayzata Wetland, Minnesota**

The Wayzata Wetland in Wayzata, Minnesota was recently investigated by Hickok et al., [7-49], to evaluate the interactions of stormwater runoff in a wetland area and determine the potential usefulness of such areas in management of water quality. The study site was a 3.06 ha peat wetland, located in the center of the City of Wayzata (population 4,500), on the outskirts of the Minneapolis-St. Paul metropolitan area. The contributing watershed (26.3 ha) consisted of mixed urban, sparsely developed, and open wooded land uses. Inflow to the wetland consisted of direct precipitation (35 percent), surface runoff (47 percent) and ground-water inflow (18 percent). Water losses were through evapotranspiration (25 percent) and surface discharge (75 percent).

- **Palo Alto Marsh/Flood Basin, California**

The Palo Alto Flood Basin is a marsh area adjacent to San Francisco Bay, which serves to impound stormwater runoff and provide flood protection to the low-lying areas of the city of Palo Alto. Constructed in the mid-1950's, the basin is formed by a series of levees and tide gates surrounding a pre-existing bayside marsh area. It receives drainage from a highly urbanized watershed of about 72 sq km. The marsh contains a mixture of salt and freshwater vegetation and upland grasses. The flood basin has been the subject of a recently concluded EPA-sponsored investigation intended to evaluate changes in surface runoff quality as a result of ponding and biological activity in the marsh [7-50].

- **Lake Tahoe Meadowlands, California**

Morris et al., [7-51] recently conducted a one-year field investigation of the effectiveness of natural marsh and meadowlands to provide treatment of surface runoff in the Lake Tahoe Basin. Seven systems consisting of four streams and three tributary drainage areas were investigated. The contributing watersheds consisted of a mix of urban, rural residential, pasture and forested lands. The primary focus of the study was on nutrient and sediment loadings associated with land development, road construction and other soil disturbance activities.

- **Cypress Stand-Orlando, Florida**

In connection with Section 208 water quality planning studies for Orange County, Florida, natural treatment of surface runoff through a cypress stand was investigated [7-52, 7-54]. The wetland study site is located at the University of Central Florida and serves a drainage area of 9.76 ha, of which 67 percent is an impervious area. The wetland is characterized by marsh vegetation and small cypress at the exterior, and mature cypress with intermittent ponds and sloughs in the interior. Stormwater from the campus grounds is channeled into the cypress stand where it follows a circuitous, slow-moving path, leaving the wetland by means of surface outflow, evapotranspiration or soil infiltration. Analysis of treatment effectiveness focused in this study on BOD, suspended solids, nitrogen and phosphorous.

o Montgomery Mall Lake, Maryland

Among the many stormwater detention and control facilities constructed in Montgomery County, Maryland since the early 1970's is the Montgomery Mall Lake [7-53, 7-54]. This is an offsite storage/detention pond serving a 60 ha watershed. Included in the drainage area are a large shopping mall, several apartment complexes, townhouses, a major highway and several secondary roads. The pond is used to limit peak flows and to reduce surface runoff pollutants. The pond has a 2.4 ha permanent pool with 45,600 m^3 of dead storage capacity. The pond is designed to control an inflow of .62 m^3/s and to release the volume at about .06 m^3/s. Grasses and wetland vegetation line the edge of the pond. Pollutant removal efficiencies of the detention facilities have been reported by McCuen [7-54].

Summary of Surface Runoff Treatment

Monitoring of the various wetland-stormwater systems includes a broad assortment of measurements related to:

- o hydrology;
- o climate;
- o vegetation;
- o water quality;
- o soils.

A summary of pollutant removal determinations for principal stormwater constituent is provided in Table 32 for each of the investigations described above. The following points should be kept in mind with regard to these data:

(1) Great disimilarities exist between the various systems studied in terms of wetland and surface runoff characteristics and monitoring procedures;

(2) Because the sampling periods upon which the data are based are generally very short (in most cases no more than one year), the statistics are not very reliable;

(3) Numerous sampling difficulties and complications in determining hydrologic components of some of the systems further reduce the reliability of the computed pollutant mass balances.

Despite these qualifications, certain conclusions can be drawn from these results:

(1) Wide disparity exists in nonpoint source pollution removal capacities of wetlands, particularly with regard to nutrients;

TABLE 32. OBSERVED POLLUTANT REMOVAL EFFECTIVENESS OF WETLAND-STORMWATER TREATMENT SYSTEMS

System/location	Wetland type	Hydraulic/ hydrologic factors	Application concentration	Removal efficiency	Comments and reference
Wayzata, Minnesota (Existing runoff situation)	Peatland	28.3 ha watershed 2.8 ha wetland area Inputs: Precip. - 2.38 ha·m G.W. - 1.20 ha·m Runoff - 3.19 ha·m	(Annual average)		o Nutrient discharge from wetlands related to seasons. o P seems to be limiting plant nutrient. Microbial activity appears to be initial and most important mechanism for P removal o Heavy metal inflow concentrations and removal efficiencies varied according to contributing land uses. Greatest reduction values for commercial area runoff. (7-49)
			NH_3-N 3.94 mg/l	Net increase	
			Total P .92 mg/l	78%	
			SS 701 mg/l	94%	
			Cd .4-1.4 µg/l	25-80%	
			Cu 12-19 µg/l	73-83%	
			Pb 26-71 µg/l	90-97%	
			Zn 10-15 µg/l	78-86%	
Palo Alto, California (Flood control basin)	Brackish tidal marsh	248 ha marsh 72 Km^2 watershed 224,000 m^3 dead storage Mainly channelized flow, high marsh areas innundated infrequently.	(Average)		o Inflow/outflow pollutant relationships extremely variable from storm to storm. o Outflow SS exhibited less variability than inflow levels. Because of hydraulics, very little solids deposition appears to occur in high marsh areas. o Heavy metal inflow and outflow levels were at limits of detectability. o Consistent decrease of Pb measured across marsh. (7-50)
			Total N 3.67 mg/l	37%	
			Total P 0.36 mg/l	Net increase	
			SS 290 mg/l	87%	
			VSS 62.9 mg/l	85%	
			BOD 12.2 mg/l	54%	
			Cd 0.05 mg/l	NA	
			Cu < 0.01 mg/l	NA	
			Ni 0.07 mg/l	NA	
			Pb 0.16 mg/l	NA	
Lake Tahoe, California (Natural system)	High altitude meadows	Several sites investigated with watersheds of several hundred to 14,350 ha. Wetland slopes of 2-7%.	NH_3-N .02-.44 mg/l	Up to 67%	o Extreme variability in nutrient removal. Many instances of increase through wetland. Most significant reductions during storm episodes. o TKN released from wetland systems during spring snowmelt runoff. o SS removal greatly enhanced by sheet flow conditions. (7-51)
			NO_3-N .02-.57 mg/l	Up to 96%	
			TKN .2-6.6 mg/l	Up to 76%	
			Total P .018-1.9 mg/l	Up to 93%	
			SS 1-2,978 mg/l	Up to 99%	

TABLE 32. (continued)

System/location	Wetland type	Hydraulic/ hydrologic factors	Application concentration			Removal efficiency	Comments and reference
University of Central Florida	Cypress stand	9.76 ha watershed, with 67% impervious area.	Total N	45	Kg/yr.	95%	o Slow moving, circuitous flow allows particulate matter to settle and accumulate rapidly.
			Total P	8	Kg/yr.	97%	
			SS	22,580	Kg/yr.	99%	
			BOD_5	145	kg/hr.	89%	(7-52, 7-53)
Montgomery Co., Maryland	Wetland detention basin	60 ha watershed 2.4 ha permanent pond with 45,600 m^3 dead storage capacity	NH_3-N	3	g/s	99%	o Large permanent pond volume is key factor in high pollutant removal efficiencies.
			Total P	0.3	g/s	99%	
		Depth - .9 to 4 m	Ortho P	0.13	g/s	93%	
		22 hr. detention time @ peak inflow rate of 0.62 m^3/s.	BOD_5	5.2	g/s	97%	
			Cd	0.18	mg/s	98%	
			Fe	4.7	mg/s	96%	
			Pb	0.18	mg/s	96%	
			Zn	0.23	mg/s	99%	(7-53, 7-54)

(2) Greatest consistency in pollutant reduction appears to be for BOD, suspended solids and heavy metals;

(3) Seasonal factors can have a major influence on pollutant removal capabilities of certain wetlands.

In addition to the tabulated data, the various investigations produced a number of findings and observations which provide greater insight into stormwater treatment effectiveness of wetland systems.

The nature of the flow through wetlands is considered to be a critical factor affecting treatment removal. In the Lake Tahoe study, the extent to which runoff followed a sheet flow pattern was found by the investigators to be the most significant factor distinguishing pollutant removal effectiveness at different meadowland sites [7-51]. The slow-moving, circuitous flow paths through the cypress stands in Florida are, likewise, a major contributing factor to high levels of pollutant reduction [7-53]. In the Palo Alto study, widespread flooding did not occur during the investigation. Rather, the flow remained confined to the well-defined channels traversing the marsh [7-50]. The resultant effect on pollutant removal is unknown but probably of significance.

The influence of the seasons and individual storm episodes was observed in these investigations. Nutrient discharges in particular were related to the seasons in the Wayzata wetlands [7-49]. The flushing and leaching effects of spring snow melt appeared to be the cause for higher total Kjeldahl nitrogen and organic carbon discharges in the Lake Tahoe meadowlands [7-41]. In terms of treatment, Morris et al., [7-51] found the most dramatic nutrient and sediment reductions in effective meadow systems to be associated with individual storm episodes.

In the Wayzata study, microbial activity in the wetlands appeared to be the initial and most important mechanism for removing phosphorus from the soil-water solution [7-49]. Under submerged, anaerobic conditions, this activity was found to decrease dramatically, suggesting a similar decrease in phosphorous removal capability.

In the Lake Tahoe investigation, the extreme variability observed, in terms of nutrient removal, strongly points to the need to carefully examine each individual wetland system, prior to predicting its usefulness in surface runoff management. In addition to flow regime, increasing slope of the wetland and extensive grazing appeared to have negative impacts on treatment effectiveness. The presence of beaver dams on the other hand seemed to have a beneficial effect [7-51].

Studies of wetland soils and vegetation provided interesting results in the Palo Alto Marsh investigation. Lead, zinc and copper concentrations in surface soils were found to be directly proportional to pH [7-50].

With changes in pH from 5 to 6, adsorption of lead, zinc and copper to soil surfaces was found to increase. Pickleweed, which predominates in the lower-elevation areas of the marsh, demonstrated the ability to extract heavy metals from the flood basin to a greater extent than the mixed marsh vegetation or upland grasses. In particular, aboveground plant tissue was found to accumulate zinc and cadmium at significant levels beyond soil and stormwater concentrations.

Extensive assessment of associated environmental impacts was not undertaken as part of these investigations. Limited field evaluation of biological (wildlife and vegetation) impacts revealed no adverse effects in either the Wayzata Wetlands or Palo Alto Marsh studies [7-49, 7-50]. The concentrations of heavy metals in soils did not appear to have substantial effects on plant conditions in the Palo Alto Marsh [7-50].

UPLAND RETENTION AND TREATMENT OF SURFACE RUNOFF

The practices of utilizing vegetated detention basins, grassed drainage swales and other upland areas for the control of stormwater runoff have gained wide acceptance and attracted considerable research attention in recent years [7-55]. Most of the literature addresses hydrologic and operational aspects of such practices. Poetner [7-56] prepared a comprehensive review of stormwater detention practices in which numerous case studies are used to illustrate various design approaches and state-of-the-art technology. Additionally, broad coverage is given to legal, environmental, economic, and operation and maintenance aspects. The potential role of detention facilities for treatment purposes is addressed primarily in terms of erosion and sediment control. A number of others have devoted research to hydrologic analysis and optimization of detention facilities [7-57, 7-58, 7-59].

Very little information has been developed with respect to the stormwater pollutant removal capabilities of grassed waterways and other upland retention and detention facilities. As a consequence, concepts and findings related to land treatment of wastewater have been used to evaluate expected treatment effectiveness and design requirements [7-60].

A number of recent 208 water quality management studies have examined various types of upland detention storage and runoff control strategies for management of nonpoint source pollutants. In one such study, Biggers et al., [7-61] used simulation techniques to evaluate projected pollutant removal effectiveness of stormwater retention and detention practices. Pollutant removal rates were approximated based upon those reported in the literature for land treatment operations relying upon rapid infiltration and high-rate irrigation techniques. The simulation exercise yielded ranges of expected annual pollutant reductions for different contributing land uses as shown Table 33.

TABLE 33. SIMULATED ANNUAL POLLUTANT REMOVAL RATES FOR DETENTION BASIN AND ON-SITE RETENTION/DETENTION CONTROLS[a]

	Removal of pollutant, %					
Management practice	Total P	Total N	BOD	Lead	Zinc	Sediment
Detention basin	33-48	21-36	31-47	82-88	47-53	88-94
On-site retention/ detention controls	52-68	37-45	73-90	86-95	85-95	89-99

a. Source: reference 7-61.

The results also pointed to hydrologic design modifications that could significantly improve pollutant removal in detention basins which have traditionally been designed as peak flow attenuation facilities. Others have also identified the need to integrate flood, erosion and pollution control technologies to benefit from the multiple uses of stormwater detention practices [7-62, 7-63].

Two recent field studies provided useful data and evidence of surface runoff pollutant control, achieved through the application of upland vegetative practices.

Retention/Detention Controls

In Orange County, Florida, various nonstructural and low-technology facilities have been installed and tested for controlling and treating stormwater runoff near its source [7-52, 7-53]. The individual unit facilities include:

(1) Percolation facilities, which provide for infiltration, detention, and overland treatment of runoff;

(2) Underdrains, providing for collection and drainage of stormwater after passage through permeable surface soils;

(3) Residential swales, providing for on-site percolation of ponded stormwater in grassed depressions;

(4) Detention/sedimentation basins, equipped with debris traps, and baffle arrangements.

The major stormwater pollutants of concern in Orange County have been identified as BOD, suspended solids, total nitrogen and total phosphorus. The implemented control measures have demonstrated their effectiveness in reducing the loadings of these pollutants to receiving waters. The combined effect of these practices has not been determined, although it is expected to be substantial. Table 34 provides the annual pollutant loadings and removal efficiencies for these various management practices. The high removal rates reflect the combined processes of detention, percolation and soil adsorption and filtration. The actual role of vegetation is not known.

Grassed Waterways

The use of grassed waterways has been, primarily, in conjunction with agricultural practices. Much of the associated research and development work has focused on hydraulic design, channel stability and erosion control functions [7-64, 7-65]. Specifications for design, construction and maintenance are included in several publications issued by the USDA Soil Conservation Service [7-65]. The emerging concern regarding use and impacts of agricultural chemicals has given rise to investigations into pollutant removal capabilities of vegetated areas adjacent to cropland. One such study was conducted by Asmussen et al., [7-66] at Tifton, Georgia. The objective of this particular study was to evaluate the capacity of a grassed waterway for removing the herbicide 2,4-D from agricultural runoff.

The waterway studied was 24.4 m long with a 2 percent slope, and was planted with bermudagrass, bahiagrass and other common grasses. Simulated rainfall (25.4 cm/hr) was used to generate runoff under dry and wet antecedent soil conditions in both the cropped area and the waterway. Sampling along the waterway showed substantial reduction in sediment and herbicide loadings as evidenced by the data in Table 35.

TABLE 35. REDUCTION IN WATER, SEDIMENT AND 2,4-D IN THE WATERWAY[a]

(TIFTON, GEORGIA)

	Reduction, %	
	Dry condition	Wet condition
Runoff (water)	50	2
Sediment	98	94
2,4-D in water phase	71	69
2,4-D in sediment phase	>99	>99
Total 2,4-D	72	69

a. Source: reference 7-66.

TABLE 34. PERFORMANCE OF RETENTION/DETENTION CONTROLS, ORANGE COUNTY, FLORIDA[a]

Stormwater control measure	Storms sampled	Influent load, Kg/yr				Effluent load, Kg/yr				Ave. removal, %			
		BOD_5	N	P	SS	BOD_5	N	P	SS	BOD_5	N	P	SS
Swale/ percolation	4	101	32.84	9.69	2,764	7.42	4.28	0.78	108	93	90	92	96
Underdrain	4	118	24.09	6.53	5,206	5.31	11.02	0.49	63	96	54	93	99
Residential swale	2	34	6.96	1.89	1,504	21.17	5.83	2.00	164	38	16	0	89
Detention/ sedimentation	6	193	59.11	21.76	8,112	77.66	14.23	5.24	1,332	60	76	76	84

[a] Source: references 7-52, 7-53.

The cover of grass, weeds and organic debris was found to be extremely effective in intercepting and adsorbing most of the 2,4-D discharged from the cropped area. This occurred in spite of the fact that (a) high rates of artificial rainfall were used, and (b) 2,4-D is reportedly very mobile in soils in comparison to other pesticides. The study suggests that more strongly sorbed compounds would be even more effectively removed by grassed waterways. Overall, diversion of agricultural runoff from cropped areas through grassed waterways or adjoining vegetated areas can provide for substantial reduction of agricultural chemical discharges to streams.

A problem associated with storm runoff and flood water can be the entrainment of high levels of sediment. In a study conducted at the Institute of Water Utilization (now the Water Resources Research Center), at the University of Arizona, two experimental sites near Safford, Arizona, were evaluated to determine the effectiveness of grass filters for sediment reduction in flood water destined for artificial recharge facilities [7-67].

One site at Safford Agricultural Experiment Station was studied for a period of three years from 1961 to 1963. Seven experimental border checks were established on a 1.84 ha site of the field station, having an average slope of 0.10 percent. Six of the plots were seeded or planted with two types of bermudagrass and other grasses. The seventh plot was seeded with Lahontan alfalfa. The second experimental site was studied in 1963. Two border checks were established on a 1.2 ha site, which had an average slope of 0.60 percent. Coastal and common bermudagrass were planted on the two plots. Grass filtration was found to be extremely effective in removing up to 99 percent of the sediment entrained in the test water with the two varieties of Bermudagrass tested at the Safford test site.

The study found that grass filters promote mechanical sedimentation by retarding the flow velocity and consequently, enhancing particle settling. A second possible mechanism for sediment removal, which constitutes a much smaller effect, is the adsorption of the collodial-size particles to grass surfaces. On the basis of the experimental results at Safford, it appears that grass filtration is an effective, economical, first-stage procedure for reducing sediment in flood water.

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SECTION 8

GUIDELINES FOR WETLAND SELECTION AND DESIGN

INTRODUCTION

This section presents basic guidelines for use in designing or selecting wetlands for water pollution control. The review of case studies and removal mechanisms makes it all too clear that a universally applicable theory of design for wetland treatment systems is not yet available. Although there appear to be trends and repeatable results in pollutant removal and wetland response, enough exceptions occur to indicate that not all relevant factors are well understood. Major differences in geographic location, climate, hydrologic regime, and type of wetland appear to significantly affect wetland response to pollutants. Thus, although there are 19 case studies reviewed in Section 7, only a few pertain to any single physical/ecological situation.

This section presents general principles that appear to reflect performance or conditions reported for most wetland systems. For more detailed guidance, based upon past experiences with a specific type of wetland in a specific climate, the original case studies reviewed in Section 7 should be consulted.

SELECTION OF WETLAND

In the development of a wetland treatment system, two major choices must be made. The first is selecting a natural versus a constructed wetland. Some researchers recommend that constructed wetlands be used in preference to natural wetlands [8-1, 8-2, 8-3, 8-4]. This recommendation appears to reflect a desire to protect natural wetlands until the mechanisms of pollutant removal and assimilation are better understood. Natural and constructed wetlands have advantages and disadvantages, the importance of which will depend upon individual circumstances.

Some advantages of natural wetlands include:

- immediate availability - no extended construction or vegetation establishment period should be necessary;
- no new land requirements.

Disadvantages of natural wetlands could include:

- o inconvenient location;
- o connection to critically sensitive water bodies;
- o inadequate size to assimilate waste load;
- o it may be required by regulatory authorities that discharges to natural wetlands meet stringent, "polished" influent criteria;
- o operation of a managed wetland may alter ecosystem balance and relationships.

To a large extent, the advantages of a constructed wetland complement the disadvantages of a natural wetland. These advantages include:

- o flexibility in site location;
- o optimum size for projected waste load;
- o construction of topographic features such as channels, shallow bars, islands and levees to improve pollutant removals and facilitate maintenance;
- o possible exemption from rigorous influent criteria if wetland is considered part of a treatment system;
- o would supplement existing wetlands;
- o ecosystem construction of a managed wetland could not be judged an alteration of a natural wetland ecosystem.

Concomitantly, there are disadvantages to a constructed wetland, including:

- o cost and availability of suitable land;
- o construction costs for grading and planting the site;
- o unavailability of the system during the construction period;
- o possibly reduced performance during vegetation-establishment period.

If a constructed wetland is selected, a second choice, the type of wetland that will be used, has to be made. Assuming that the necessary geologic and hydrologic conditions can be found, ponded or grassed wetlands can be constructed almost anywhere. However, certain types of wetlands would probably not become successfully established outside of their current geographical distribution. Cypress swamps will not flourish in Wisconsin and peat bogs are not feasible for Arizona. In addition, wetlands planted with particular plant species, unless rigorously managed, will become dominated by plant species tolerant of local hydrologic and climatic conditions.

Nonetheless, sufficient choices among marshes, open ponds and forested land remain and are discussed in more detail later in this section.

DESIGN FACTORS

The factors affecting wetland performance and basic principles of design can be used to plan a constructed wetland. These same factors and principles can be used to assess an existing wetland for its suitability as part of a treatment system. It must again be emphasized that local conditions greatly affect the usefulness of wetlands for water pollution control and a site-specific evaluation must be conducted.

Factors that have been identified in the case studies include:

- type of wetland;
- hydrologic system;
- vegetative system;
- seasonal factors;
- nature of polluted water;
- loading rate;
- target pollutants.

These design factors and principles or guidelines, based upon the case studies and removal mechanisms, are discussed below.

Type of Wetland

The following seven basic types of wetlands have been identified:

- northern peatlands;
- cattail/grass marshes;
- southeastern swamplands;
- cypress domes;
- freshwater/tidal marshes;
- open ponds;
- meadows and seepage wetlands.

All seven can be used for the treatment of polluted water. However, of these seven, only four are currently considered for constructed wetlands:

- cattail/grass marshes;
- freshwater/tidal marshes;
- ponds;
- meadows and seepage wetlands.

No guidance is available on the selection of one wetland over another. Local conditions and needs may be prime factors in the selection of a wetland system. Studies at the Brookhaven National Laboratory indicated that a meadow-marsh-pond system performed slightly better than a marsh-pond system [8-5, 8-6]. To the extent that different pollutants are removed by different plants and mechanisms, a wetland system that employs combinations of the wetland types should have improved overall performance.

Hydrologic System

o Polluted Water Inflow

Flow of water to a wetland has three characteristics: quantity, duration and frequency. Quantity, or loading, is discussed in the following section. Duration and frequency are the principal characteristics that distinguish municipal treatment plant effluents from stormwater. Even allowing for daily and monthly flow variations, treatment plant effluents are relatively steady. Stormwater influent to a marsh will vary in duration and intensity among storms and seasons.

Wetlands that are permanently flooded, such as cypress domes and marsh-pond systems, can readily accommodate either continuous or intermittant polluted water inflow. Such wetlands usually are connected to other sources of water, such as a river, lake or ground water, and their ecosystems are not dependent upon the polluted water flow.

Wetlands that are not permanently flooded, such as some constructed wetlands and meadow-marsh systems, may be sensitive to hydrologic changes caused by the inflow of polluted water. Constructed wetlands which rely upon continuous inflow may suffer during the dry periods between stormwater inflows. On the other hand, the ecological balance of intermittently-flooded wetlands may be disturbed when the water supply becomes continuous. In either case, the wetlands will evolve into different systems over time in response to changes in the hydrologic regime.

The selection or design of the wetland should match the expected flow of water. This is important for two reasons:

(1) Flooding of the wetland should coincide with the growth requirements of the vegetation characteristic of that wetland;

(2) Addition of pollutants to the wetland should coincide with the vegetative growth phase during which those pollutants are absorbed or metabolized by the plants. For example, removal of phosphorus was found, in a number of studies, to be a seasonal phenomenon, with actual plant release of phosphorus in winter and early spring [8-1, 8-7, 8-8, 8-9, 8-10, 8-11].

- **Inflow/Outflow Regulation**

Because of the temporal variations in stormwater runoff quality, the idea of selectively regulating the inflow to wetland treatment areas is a promising one where conditions so permit [8-28]. Work has been done in Florida to utilize infiltration areas and retention basins for containment and treatment of the "first flush" component of runoff [8-12, 8-13, 8-14]. In one instance, automatic gates were installed to control inflow to the off-channel treatment pond in accordance with streamflow conditions. This approach may apply equally well to artificial wetlands and some natural wetlands amenable to flow regulation.

It may be desirable to regulate the outflow from wetlands for flood control and water quality control. The use of dikes, weirs and other overflow structures can be employed selectively to increase wetland storage and detention time. For instance, some have suggested the retention of runoff flows during springtime flushing [8-15]. The objective would be to store runoff waters until marsh communities were functioning at high uptake rates. This management approach would have to account for hydraulic surges in runoff, and possibly provide for diversion of a portion of the flow around the wetland.

- **Seasonal Application**

Whereas very little can be done to influence the timing of stormwater discharges to wetlands, small wastewater flows may be retained in holding basins for seasonal application to take greatest advantage of biological treatment processes. The timing of discharges should take into account (a) biological activity within the wetlands, (b) availability of dilution flows (i.e., rainfall and runoff), and (c) seasonal uses and quality of waters downstream of the wetland discharge. Exempt from this approach are wastewater discharges to palustrine inflow and seepage wetlands, for which seasonal aspects generally need not be considered, providing soil conditions are suitable for phosphorus adsorption or nitrate reduction [8-15].

Seasonal flushing of wetlands is an important management consideration. Spring flushing of nutrients and organic matter can be especially detrimental to receiving waters in contributing to summer algal problems [8-16, 8-17]. In coastal areas, increased discharge of nutrients from wetlands may further aggravate red tide problems, which have been linked in some instances to the flushing of freshwater marshes [8-18].

The suggestion has also been made to use flushing flows to strip nutrients from the wetland system after they become mobilized in late autumn [8-15]. Whether or not further treatment may be required for such concentrated discharges would depend on the nature of receiving water impacts at the time of flushing. For artificial and small wetland systems, it may be feasible to divert concentrated flushing flows for final treatment by upland soils and vegetation [8-15].

o Flow Routing

The manner in which wastewater flows are introduced and distributed within wetlands is an important management consideration. Several investigators have recommended that urban and agricultural runoff waters be directed into marsh areas rather than open water bodies within wetlands [8-19, 8-20]. The objective here is to maximize the effective contact between wastewaters and wetland soils and vegetation, and avoid short-circuiting of pollutants through the system. Even greater interaction with soils and vegetation can be achieved by applying the waste flow on upland areas adjacent to wetlands [8-18]. Higher rates of nutrient removal may be attained by virtue of the greater amount of seepage and soil contact which this technique provides [8-20]. Careful consideration must be given to the scouring effects of stormwater flows in high marsh and upland areas. In addition to healthy vegetative cover, rock berms and other types of energy dissipation structures may be needed to protect these highly erodible soils.

Pollutant removal effectiveness within the wetland depends upon wide distribution and circuitous flow paths. In natural wetlands, there may be little that can be done to enhance this aspect of pollution control without altering the ecosystem. Where canals may be constructed in wetlands for purposes of navigation, mosquito control or other development activities, certain practices have been recommended to minimize disturbance of pollutant removal and other inherent wetland functions. These include:

(1) Limit construction of canals that connect the edge of the hydrological basin to the middle (shortens flow-through and detention time) [8-21];

(2) Limit construction of blind-end canals or finger-fill developments (leads to poor circulation and bypassing of flows) [8-22];

(3) Limit construction of impoundments in marsh areas (decreases marsh/treatment areas) [8-23];

(4) Allow periodic openings in dredge spoil banks so water circulation is not impeded [8-23, 8-24].

In the creation or reconstruction of wetlands, more leeway is possible in directing water flow through the system. The researchers at the Brookhaven National Laboratory strongly suggest that water be routed through a mix of wetland features (i.e., meadows, marshes, islands and open water) [8-6]. This structural diversity provides more biological niches and greater opportunity for assimilation of pollutants in passage through the wetland. A hypothetical comparison of overall removal efficiency of an unbroken versus a compartmentalized wetland, as proposed by Blumer [8-35], is provided in Figure 9. Examples of alternative wetland arrangements that might be selected to enhance the removal of nutrients and other pollutants are shown in Figure 10.

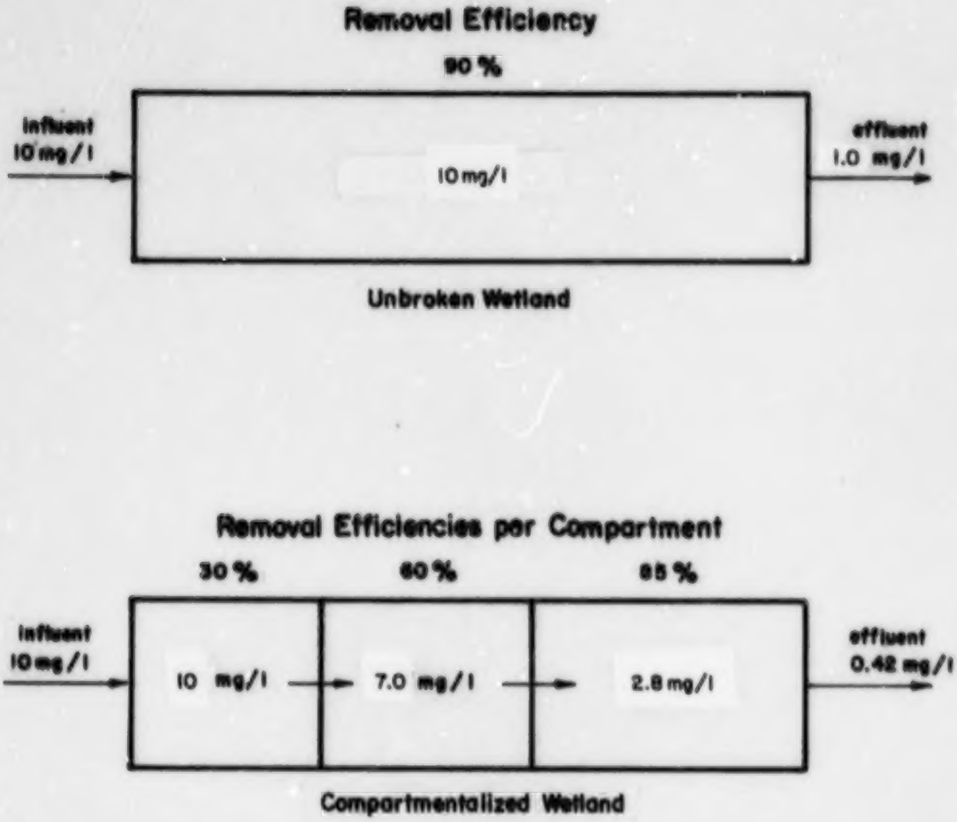
Removal Efficiency
90 %
influent
10 mg/l
10 mg/l
effluent
1.0 mg/l
Unbroken Wetland
Removal Efficiencies per Compartment
30 %
60 %
85 %
influent
10 mg/l
10 mg/l
7.0 mg/l
2.8 mg/l
effluent
0.42 mg/l
Compartmentalized Wetland

Figure 9. Comparison of renovation potential of two hypothetical wetland systems.[a]

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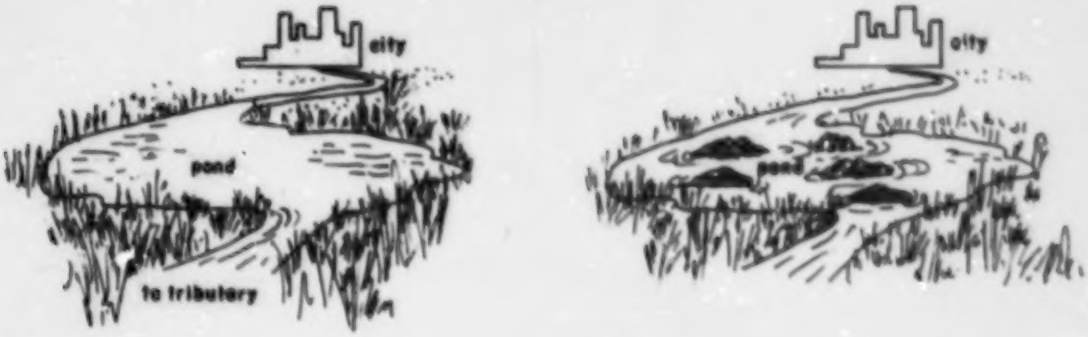
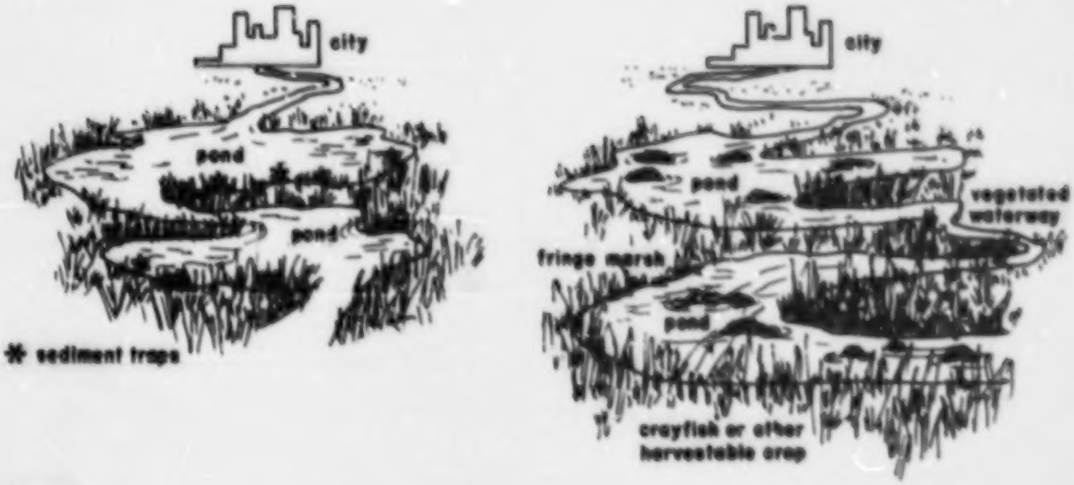
a. Source: reference 8-25.


city
pond
to tributary
city
pond

Wetland with straight-through flow least desirable.

Island wetland meadows with increased trapping surface and shape intercepts flow and sediments.


city
pond
pond
* sediment traps
city
pond
vegetated waterway
fringe marsh
pond
crayfish or other harvestable crop

Wetlands placed in meandering sequence. Flow forced to migrate around edges.

Wetland drainage channels with intermittent and terminal ponds.

Figure 10. Alternative wetland arrangements designed for increased nutrient retention which might be possible in constructed wetland areas.

o Water Level Maintenance

Water levels within wetlands are closely related to pollutant removal functions and flow distribution. Manipulation of water levels through the use of weirs, dikes, flow regulation, etc., must take into account the competing needs and resultant effects in different portions of the wetland. For example, best pollutant removal efficiencies are reportedly achieved with shallow water depths [8-26]. To induce shallow flow in meadows and high marshes may require impoundment and greater depth of flow in lower ponds and channelized sections of the wetland. This alteration in flow regime is likely to increase sedimentation rates, but may bring about undesirable changes in soil and vegetative uptake of pollutants in the affected open-water areas. Clearly, tradeoffs must be made to obtain the optimum pollutant control effectiveness through water level manipulation in wetlands. This must proceed from a very thorough understanding of the interrelationships of flow regime and ecosystem functions throughout the entire wetland.

o Infiltration

Wetlands management for pollutant removal should focus on providing the greatest amount of soil-water contact. Seepage of ground-water influent systems are most desirable because of the opportunity for wastewater purification within the soil mantle. The cypress wetlands of Florida provide ample evidence [8-27, 8-28]. Hydraulic routing of wastewaters and runoff should take maximum advantage of permeable soil areas within wetlands to provide infiltration. Also, sandy soils may be imported to form filtering devices in channelized sections [8-25].

Construction of artificial seepage wetlands has met with success in some instances. The objective is to create conditions where vertical and lateral water movement is slow enough to maintain high soil moisture and allow wastewater renovation, yet rapid enough to prevent surface overflow [8-29]. Sutherland has found the most useful types of soils to be silty in composition with hydraulic conductivities in the range of 10^{-4} to 10^{-5} cm/sec. [8-29, 8-30].

Vegetative System

The vegetative system in a wetland is a function of climate, hydrologic factors and the availability of nutrients. In a natural wetland, the vegetation will be established, and is a factor in the selection of loading rates and expected pollutant removal. The responses of various plant species to polluted water were discussed in Section 5.

In addition to the efficiency of pollutant removal and plant species tolerance to pollutants one other factor is important. Polluted water inflow to a wetland, whether treated municipal wastewater or stormwater, represents a new source of increased nutrients. This will alter the ecological balance previously existing in the wetland and may result in a substantial change in relative plant population. The wetland's response to this change may be negative. However, the newly dominant plant species, if any, presumably emerge because of their ability to use the new nutrient source (wastewater), or because of their tolerance to the pollutants. Such a system response should therefore be beneficial to the objective of pollutant removal.

In the construction of wetlands, the selection of vegetation should be guided by climate, hydrology and the response of plants to pollutants. The plants discussed in Section 5, such as Elodea, duckweed, water hyacinth, cattail and sedge, each have a particular environmental requirement for growth. It should be safe to assume that use of these plants, in their native geographical areas, will result in their retention in the wetland. Use of these plants in areas where they are not native, may result in their ultimate replacement by local plant species.

The findings presented in Section 7 indicated that diversity of vegetation, and of hydrologic regimes, improved a wetland's overall pollutant removal. Monocultures of water hyacinth have been used successfully in intensively managed aquaculture systems. However, for wetlands with diverse ecological niches, and which are intended for multiple uses, a diversity of vegetation approximating a natural system is recommended.

Section 5 presented three basic plant groups: (a) floating, (b) emergent, and (c) submerged. Each group contains species which demonstrate high uptakes of specific pollutants. The selected plant mix should match the plant species with the pollutants of concern.

Vegetation also serves as a filtration system, and a support medium for microorganisms which metabolize organic substances, and nitrify ammonium. To optimize the microbiological removal mechanisms, vegetation that forms dense stands of submerged stems and leaves, or has heavy, thick floating root mats should be selected.

- **Establishment of Vegetation**

Whereas landforming for the creation of channels, ponds and islands represents a simple application of common dredging and grading techniques, establishing vegetation in a constructed wetland is an unfamiliar procedure to most engineers.

New vegetation may be established in a constructed wetland from seed or transplants, or by natural establishment. Commercial nurseries generally do not carry seeds or plant materials common to wetlands. The best source is a wetland similar to the constructed system. When a constructed wetland is near a natural wetland, there will be movement of seeds and plants to the constructed system. The East Bay Regional Park District in California uses transplanted material to the extent permitted by funds [8-31]. In the establishment of new or reconstructed marshes, cattails have been planted over 5 to 50 percent of the suitable land area. The vegetation has then become fully established within three to five years.

Seasonal Factors

The seasons are an important influence on all wetlands. This is a natural consequence of reduced biological activity.

Most case studies reviewed in Section 7 identified seasonal variations in pollutant removal. In northern climates with freezing winter temperatures, removals of BOD and phosphorus substantially decreased during the cold season and in warm-climate wetlands, annual dormancy of some vegetation produced similar decreases. In contrast, summertime algal production increased effluent BOD and suspended solids at the Martinez, California marsh [8-32, 8-33].

The following guidelines for dealing with seasonal effects on wetlands are proposed:

(1) Physical pollutant removal processes, such as sedimentation or adsorption on soil or peat, are less likely to be affected by the season. Wetlands relying on these processes can be designed for year-round operation.

(2) Biological removal processes, such as for nutrients or BOD, can be substantially affected by the season, particularly in cold-weather climates. Low winter temperatures may kill some plant species and will lower the metabolic rate, and hence the pollutant assimilation rate, of bacteria. When poor removal or poor assimilation of pollutants is unacceptable, alternative discharge locations or seasonal storage of the polluted water must be provided.

(3) Deep water systems such as ponds or swamps are less likely to be affected by cold weather than shallow ponds or meadows, where influent can build up in frozen layers over the winter, and be discharged almost untreated in the spring.

(4) Where seasonal plant dormancy can adversely affect a constructed wetland's pollutant removal efficiency, providing a diversity of vegetation will minimize the deficiencies of any single species.

Nature of Polluted Water

Two basic groups of polluted water can be considered for treatment by wetlands. Treated municipal wastewaters can be discharged to wetlands as a polishing step, to further remove relatively low levels of polluting constituents. Urban runoff, which, as discussed in Section 3, contains substantial amounts of pollutants, can be discharged through wetlands to protect a more sensitive or valuable receiving water.

As discussed under Polluted Water Inflow, wastewaters differ from stormwaters in their flow. As noted in Section 3, they also differ from stormwaters in their constituent concentrations. The concentration of treated wastewater pollutants applied to wetlands, as reported in the case studies in Section 7, are presented in Table 36.

TABLE 36: APPLIED POLLUTANT CONCENTRATIONS

	Treated wastewaters, range mg/l		
Pollutant	Marshes	Hyacinth ponds	Experimental
BOD_5	19-44	19	110-173
Suspended solids	22-154	46	97-353
Total Kjeldahl nitrogen - N	4-23	4	12-25
Ammonium nitrogen - N	7-11	2	8
Total phosphorus - P	0.4-11	15	4-7

The value of determining influent pollutant concentrations is to help establish a correlation with pollutant removals. The wetlands reported in the case studies achieved removals of all pollutants, though not always year-round. No correlation between concentrations and removal performance was noted among the case studies, indicating other overriding factors. Stowell et al., studied the removal of BOD, suspended solids and phosphorus in relationship to influent concentrations or loading in water hyacinth and marsh systems [8-35]. Their data indicate BOD_5 removal in proportion to loading (Figures 11 and 12); however, the relationships between loading and suspended solids and phosphorus were unclear.

A correlation between influent pollutant concentrations and wetland damage or stress may be hypothesized. System stress, with marked reduction in species diversity, was noted at Hay River [8-34]. However, a comparable concentration of pollutants were discharged at Great Meadows [8-10] with no apparent system stress, and even higher levels were accommodated in the artificial Brookhaven National Laboratory system [8-5, 8-6].

With no clear rationale for matching influent water quality to a wetland, and with several years of research still to be conducted before sufficient information is collected, caution should characterize the application of polluted waters to wetlands. In general, the following application guidelines are recommended:

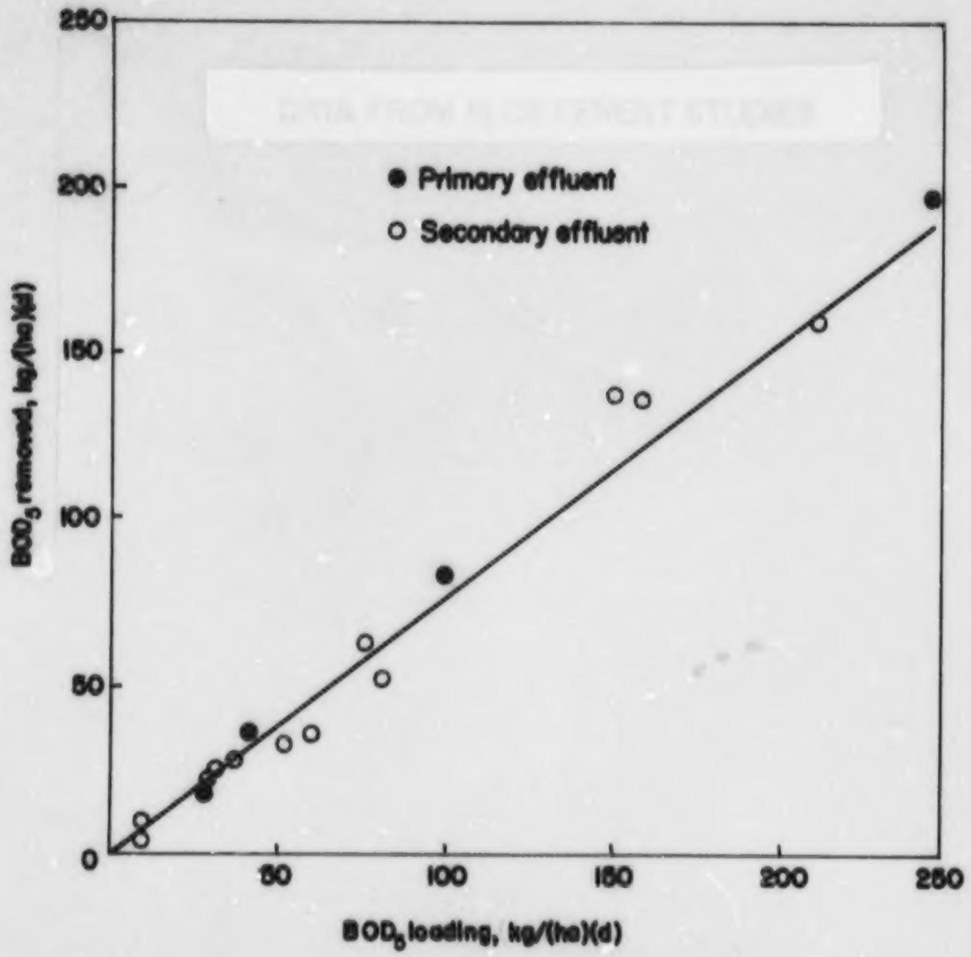


Figure 11. Effect of BOD loading on BOD removal in water hyacinth systems[a] (data summarized from 16 studies).

a. Source: reference 8-35.

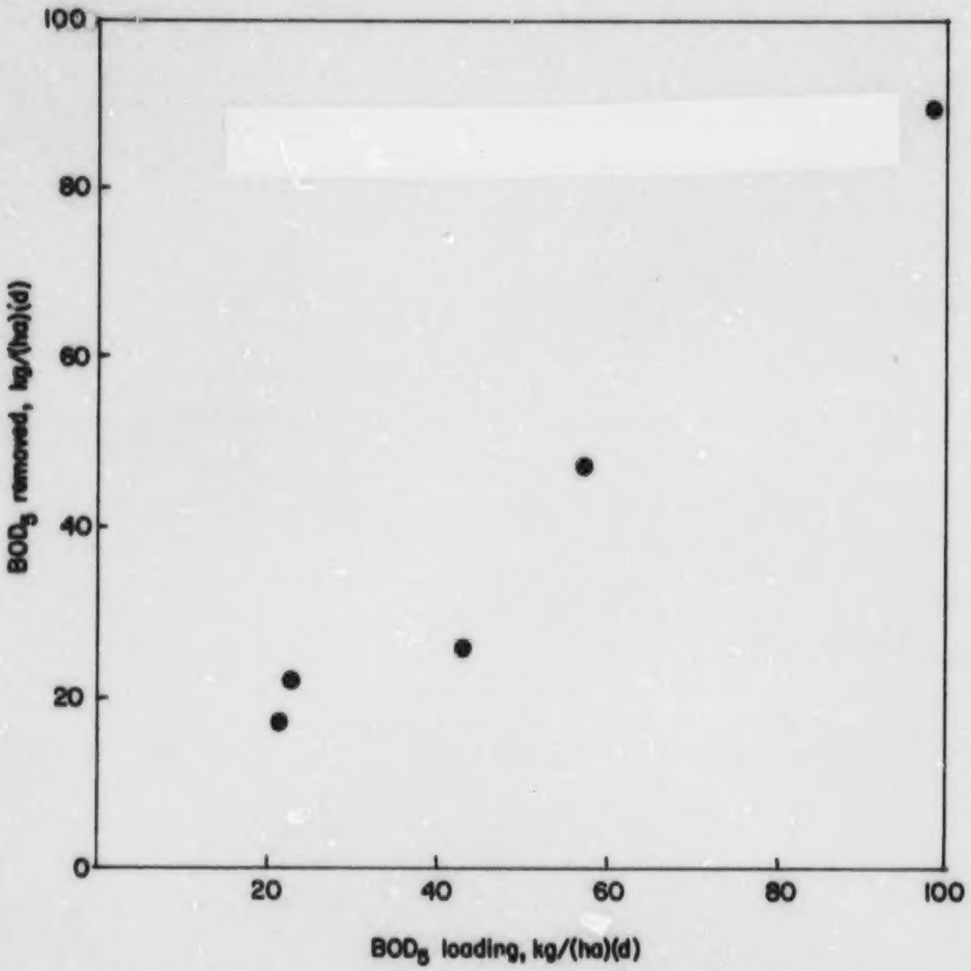


Figure 12. Effect of BOD loading on BOD removal in marsh and peatland systems[a] (data summarized from 5 studies).

a. Source: reference 8-35.

(1) Natural and constructed marshes - secondary level treated municipal effluents and stormwaters;

(2) Water hyacinth ponds - secondary to primary level effluents and stormwater;

(3) Intensely managed constructed systems - primary level treated municipal effluents must still be considered experimental.

Loading Rate

The loading rate of a polluted water upon a wetland should correlate with pollutant removal and system response. Two types of loading are considered: (a) hydraulic loading, and (b) application of individual pollutants. Wetlands performance and response (health of the ecosystem) are also determined by type of wetland, vegetation, hydrologic factors, climate and season. Thus, it is difficult to establish firm relationships between loading and performance from the few case studies. The problem is compounded somewhat by differences in measurement techniques and reporting of the parameters of concern.

Table 36 summarizes reported loading rates for the case studies in Section 7. The selected loading rate for a planned discharge may be compared to those in Table 36, but should be evaluated for the specific circumstances of the project.

Target Pollutants

One objective of any wetland treatment system will be the alleviation or prevention of a water pollution problem. The problem will usually be associated with one specific pollutant or a set of pollutants, such as oxygen-demanding organic wastes, nitrogen, phosphorus or heavy metals. The selection or design of a wetland should provide the removal mechanisms (physical or biological) effective for the pollutants of concern. These mechanisms have not been completely documented and their interrelationship is not fully understood. However, certain relationships can be inferred from the case studies and from research on individual plant species.

Sections 4 and 5 presented in great detail the physical, chemical and biological pollutant removal mechanisms in a wetland. The processes are complex and do not act independently. Vegetation enhances sedimentation of suspended and colloidal pollutants; sedimentation makes available many pollutants for use by vegetation; and certain plant species are substantially more effective than others in removing specific metals or nutrients.

It should be noted that the relationships presented below are subject to confirmation by further studies of wetland systems. However, to the extent that one or several important pollutants are best removed by a specific process, that process should be emphasized in the wetland, as in the following examples:

(1) Meandering channels, with slow-moving water and large surface areas, enhance settleable pollutant removal by sedimentation.

(2) Seepage wetlands or shallow flow regimes are effective for removal of pollutants such as phosphorus and metals by adsorption to the soil.

(3) Seepage wetlands, meadows and thickly vegetated wetlands are particularly useful for filtering colloidal suspensions and where filtration is important.

(4) Rapid plant growth, generally associated with harvesting, optimizes nutrient removal. For such applications, a monoculture system, such as hyacinth ponds, can be very effective. However, large amounts of vegetation must be harvested. Stowell estimates 7 kg of hyacinths may need to be harvested per m^3 of wastewater treated [8-1].

(5) Nitrogen removal by denitrification will occur in anaerobic bottom sediments common to wetlands. Deep areas, where sediments and organic detritus can accumulate in an anaerobic environment should be designed into a wetland intended for nitrogen removal.

(6) BOD removals in wetlands are accomplished by microorganisms. Optimum BOD removal will be achieved where there is high surface area (soil, plant stems, leaves and roots) for microbial growth, uniform distribution of the BOD load, and adequate dissolved oxygen. Open water surfaces in the wetland will increase oxygen transfer to the water.

(7) Because many plants are selective in their accumulation and biomagnification of various heavy metals, mixed stands of vegetation may provide the best overall heavy metal removals.

(8) As demonstrated in the Brookhaven National Laboratory studies [8-5, 8-6], varied or mixed wetland systems containing features of ponding for sedimentation, shallow areas for adsorption by soil, and mixed vegetation, have high potentials for treating typical primary level municipal wastewaters with significant concentrations of many pollutants.

Contingencies

Wetlands receiving either treated wastewater discharges or urban stormwater may be subjected to transient high loadings of some pollutants. This may be due to treatment plant failure, unusually high soil erosion in the watershed, or a chemical spill in the watershed. Contingency plans and special facilities should be prepared for natural and constructed wetlands. The contingency may be simply an emergency bypass, if the ultimate receiving water can accept the degraded discharge. Where this is not possible, and the wetland may be harmed by overloading, storage of the influent might be appropriate.

For a new constructed wetland in the City of Fremont, California, ABAG has included a protection measure as part of the preliminary marsh design. A sediment trap will be included between the urban stormwater conveyance channel and the marsh. This would intercept heavy loads of sediment expected from new construction in the watershed. The discharge from the sediment trap to the marsh will be by submerged conduits with lift gates. In the event of an oil or gasoline spill in the watershed, the trap should serve to capture the floating material.

It is beyond the scope of this report to develop contingency measures. However, it is recommended that any wetland used for pollution control be provided some measure of protection against accidental releases of damaging substances.

SECONDARY ENVIRONMENTAL IMPACTS

The primary impacts of polluted water discharge to wetlands have been discussed and include:

- pollutant destruction;
- pollutant accumulation in sediments and plants;
- possible changes in the balance of the plant community.

Secondary impacts that may be associated with the use of wetlands for pollution control include:

- disease vector organisms;
- nuisance organisms;
- odors;
- fog generation.

These secondary impacts are discussed below.

Disease Vector Organisms

Mosquitos are the disease vector organisms associated with natural or constructed wetlands. Marshes may produce several species of mosquitoes. Some species cause nuisance problems, while others may transmit disease, constituting a public health hazard. Throughout the U.S., various researchers and public health agencies have devoted considerable effort to investigation of mosquito control [8-36, 8-37, 8-38, 8-39, 8-40]. The following marsh conditions have been found to deter or limit access by natural predators and thus be conducive to the uncontrolled production of mosquitoes:

- the presence of emergent, submergent and floating vegetation;
- flotage (debris) on the water surface;
- extensive shallow areas;
- irregular margins of the marsh;
- poor water quality;
- periodic water-level fluctuations.

The primary management tools that have been promoted for mosquito control in marshes focus on natural predation (Gambusia affinis, mosquito fish), channelization and water level manipulation [8-41, 8-42]. Table 37 provides guidelines established by the Alameda County (California) Mosquito Abatement District for marsh and mosquito management [8-42]. The guidelines are designed to enhance natural predation by fish on mosquito populations, while at the same time avoiding conflict with the objectives of marsh planners.

Nuisance Organisms

Nuisance organisms may be natural inhabitants of a wetland that are a nuisance due to their proximity to urban areas. Such organisms include various species of wildlife that could be considered desirable indicators of a healthy, "natural" system. If necessary, this wildlife can be trapped and relocated to other areas.

Some nuisance organisms can be detrimental to the wetland treatment system. Specifically, the burrowing of muskrats can weaken dikes and levees. If these structures are essential features of the wetland, the muskrats must be removed from the sensitive areas.

There is also a risk that plant species introduced into wetland systems may become nuisances. Water hyacinth is a nuisance plant in many parts of the southern United States, where it clogs waterways and water supply structures. Where the growth of such a nuisance plant cannot be tolerated, then the selected wetland treatment system should employ other plant species native to the area.

Odors

Odors from wetland treatment systems are associated with anaerobic decomposition on and in the sediment. They can be controlled by assuring an aerobic water layer over the sediment. Where a wetland is heavily loaded with organic wastes, artificial aeration may be needed,

TABLE 37. RECOMMENDATIONS TO PREVENT MOSQUITO PROBLEMS ASSOCIATED WITH FRESHWATER MARSHES[a]

Marsh parameter	Recommendation	Rationale
Design of new marsh or modification of existing marsh	Provide deep water areas (1 meter or deeper)	For maintenance of fish during dry periods. The fish populations will provide the primary natural predator control for the mosquito populations.
	Avoid extensive shallow areas	Thick emergent vegetation will occur, in time, in shallow marsh areas. The effectiveness of mosquito predation by fish and other invertebrate predators could be severely limited by the presence of vegetation. It can be recommended that slopes of the margins and islands of a marsh be designed to be steep enough to reduce the volume of areas of the continuous emergent vegetation. Fish predation would be more effective if emergent vegetation areas were limited to relatively narrow margins and small islands. Marshes designed in this manner could have large amounts of vegetation, but they would have maximum interface with deep water areas where fish predators would abound.
	Avoid irregular margins	Extreme irregularity of the vegetated margins along the edge of the marsh and the islands should be avoided. Narrow coves created by irregularity of the margins would create vegetative areas with limited access for fish. Gentle curving margins would be more compatible with natural mosquito control.
	Provide sufficient quality and quantity of water	This is necessary to insure the long-term availability of an adequate supply of fish to prey upon the mosquitoes. Poor water quality may result in fish kills. Inadequate water supply may also result in fish kills if the water quantity is not sufficient to maintain the fish through the dry period.
	Establish water control structures	The ability to manage the water level should be maximized. Mosquito populations can be managed through water level management. In the event of excessive mosquito production, drawing down the water for a short period of time from the vegetation area would enable mosquito predators to prey upon the larvae in the deep water areas.
	Establish proper drainage	This insures that, in the event of a draw down, all areas drain to the deep water pools leaving no shallow isolated pools where predator fish cannot reach mosquito larvae.
	Provide access for marsh management equipment	Aquatic or terrain equipment may be necessary to accomplish control of mosquitoes or vegetation, or to plant fish. Management plans should be formulated in advance and access designed for the kind of equipment required to properly manage the marsh.

(continued)

TABLE 37. (continued)

Marsh parameter	Recommendation	Rationale
Marsh operation and long-term management	Management of vegetation	Thinning or removal of vegetation allows for more open water in vegetation stands and greater access for predator fish. Nuisance plant species can also be selectively removed.
	Dredging of main channels and ponds	Perform as necessary to remove silt deposition and maintain deep water areas as desired.
	Management of fish populations	Limit overcrowding of fish and depletion of fish food items. Transfer fish to areas with low fish populations to ensure adequate mosquito control.
	Periodic monitoring of mosquito populations	Mosquito production should be monitored to determine if immediate control measures such as water drawdown, transfer of fish populations or insecticide applications are necessary.
	Management of water levels	Water level may be drawn down to concentrate mosquitoes and fish in deep water areas; or water levels may be raised to allow fish to enter into previously shallow inaccessible areas for predation.
	Emergency insecticide applications	Insecticides should be applied as necessary to control excessive mosquito larvae. (The frequency of insecticide applications will be a function of proper design and management of the marsh. A properly designed marsh with effective management should require no insecticide applications).

a. Derived from reference 8-42.

not only for odor control, but also to assure microbial degradation of the waste. For lightly loaded wetlands, a passive aeration system would consist of open stretches of water for oxygen exchange.

Fog Generation

Generation of fog can occur when the water temperature exceeds the air temperatures and the moisture content of the air is near saturation. Fog will be a particular hazard at roads and highways. Little is known about control of fog generation in wetland systems.

Beneficial Secondary Impacts

It is becoming increasingly clear that creation of artificial wetlands holds substantial promise as a means of attaining improved water quality management in concert with other social and environmental objectives. Interest and work in the field of wetlands development is not new. Promotion and guidance have come for many years from fish and wildlife preservationist organizations and agencies. A thorough text on the subject was published in 1969 by the State of Wisconsin, Department of Natural Resources [8-43]. It covers practices related to such aspects as impoundment construction, water level manipulation, wetland farming, nesting island construction and vegetation control. The primary emphasis is on management of waterfowl habitats. Another useful reference on the subject addresses artificial wetlands development in conjunction with highway construction [8-2].

These references address many of the basic planning and design considerations pertaining to artificial wetlands-pollution control projects, insofar as maintenance of a functional wetland ecosystem is concerned. Some of the key parameters and management guidelines are listed in Table 38.

The methodology for incorporating pollution control objectives into wetland development projects is undefined at the present time. As pointed out in this section, consideration may need to be given to such factors as:

- o regional hydrology;
- o ecosystem/hydrologic responses;
- o wetland requirements for waterfowl or other environmental objectives;
- o hydrologic-pollutant removal relationships.

MAINTENANCE

Very little information is available about maintenance of wetland systems used for pollution control. Maintenance of hyacinth pond

TABLE 38 . WETLANDS DEVELOPMENT AND MANAGEMENT GUIDELINES FOR WATERFOWL ENHANCEMENT[a]

Parameter	Criteria
Size	o Watershed to wetland ratios of 20:1 (rolling hills) to 30:1 (feather terrain) are commonly recommended by U.S. SCS. Local climatic factors and watershed character may cause significant variation. o Several small impoundments have greater positive effect on waterfowl than one large marsh.
Soils	o Most desireable locations are poorly drained soils with high water table or an underlying impermeable layer. o Additions of gravel or inorganic soil to existing organic soils can improve stability for wetland vegetation.
Slope	o <1 percent wetland slope recommended.
Configuration	o Irregular shorelines offer substantially greater support for wildlife than small symmetrical impoundments.
Water depth	o Not deeper than about 4 feet for fish and wildlife needs. o Lower quality soils (in terms of productivity) should be flooded at shallower depths, with poorest soil flooded <1 foot.
Composition	o Mix of open water and emergent vegetation stands. o 50-75 percent shallow enough to achieve emergent plant growth (roughly 2 foot depth).

[a] Derived from references (8-43) and (8-44).

systems would be similar to that for waste stabilization ponds with the following additional provisions:

- periodic harvesting and disposal of excess plant material;
- protection of hyacinths from cold weather;
- maintenance of structures to retain hyacinths in the pond.

Further information on hyacinth systems is available from Dinges [8-11] and Middlebrooks [8-44].

Maintenance of marshes, seepage wetlands, swamps and bogs will be, principally, repair of channels, berms and hydraulic control structures. Harvesting of excess or dead vegetation obstructing waterways may be necessary. Disposal of large volumes of plant material may pose a problem. However, some plant material may be converted to animal feed, compost or energy. The inclusion of resource recovery operations will depend upon the economic viability of each project.

Stowell evaluated sediment accumulation in wetlands, and concluded that most bottom deposits will come from organic detritus of marsh vegetation [8-35]. A sample calculation, based upon a wetland loading of 30 $m^3/(ha)(d)$ of wastewater containing 100 mg/l of suspended solids, shows that the wetland would accumulate, at a capture efficiency of 100 percent, only 0.1 mm/y of bottom deposits due to influent solids. It thus appears that sediment accumulation would not be a major problem with most waters, and that channel dredging or cleaning may be an infrequent operation.

COSTS

Few constructed wetlands have reported construction costs, and the basis is not always comparable. Costs are highly dependent upon wetland type, size, and location. Similarly, maintenance costs are also project specific. Tchobanoglous reported a comparison in capital and operating costs between activated sludge with chlorination and primary clarification, with constructed wetlands and chlorination [8-45]. As shown in Table 39, wetlands costs are substantially lower.

TABLE 39: COMPARISON OF WETLANDS AND ACTIVATED SLUDGE COSTSa

	Activated sludge			Wetland		
Cost element, June 1979	0.1 mgd	0.5 mgd	1.0 mgd	0.1 mgd	0.5 mgd	1.0 mgd
Capital cost, $ x 10^6	0.71	1.23	1.6	0.37	0.55	0.90
O & M cost,($/y)x 10^3	35	78	117	21	48	74

a. Source: reference 8-45.

A wetlands receiving effluent from a secondary level wastewater treatment system was recently constructed by the East Bay Regional Park District in Hayward, California [8-31]. Total grading cost in 1979 and 1980 was $550,000 for an 80-hectare wetland. Because the facility will also serve as a regional recreational area, two bridges, gates, and a parking area were added for $200,000. Finally, $100,000 was reserved for transplanting vegetation to the marsh from other wetland areas.

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SECTION 9

MONITORING AND RESEARCH NEEDS

Extremely encouraging results have been reported from the studies and investigations reviewed in this report. Investigations of individual plant species have demonstrated measurable capabilities to remove pollutants from water. Similarly, physical and chemical processes known to occur in wetlands have substantial pollutant removal value. In the limited number of case studies on wetlands renovation of polluted waters, several showed excellent results. Others produced mixed results, often reflecting lack of management or improper loading of a wetland, due to lack of understanding of wetlands processes, rather than an inherent failure of the processes. Thus, if any single research objective for further wetlands studies were to be highlighted, it would be to develop an understanding of the interplay of all wetland components in the removal, assimilation, or release of pollutants. Much of the previous work on wetlands, or vegetative systems, is characterized by a piecemeal approach with little attention to system dynamics.

Much additional work remains to be performed before limited objective questions can be answered, much less a full understanding achieved about wetlands systems. Recommended areas for monitoring and research are presented in this section.

WETLANDS INVENTORY

Over the years, a number of efforts have been made by various agencies to identify wetlands. Appendix B contains a map and state-by-state listing of state and local wetland surveys conducted between 1965 and 1975 [9-1]. This information was compiled for the U.S. Fish and Wildlife Service, prior to embarking upon its current nationwide inventory of wetlands [9-2]. This massive effort is intended to identify wetlands and establish a common classification system grouping ecologically similar habitats. In relation to water quality and wastewater treatment in wetlands, this inventory will be of great aid in establishing uniformity in concepts and terminology, and ultimately in improving understanding of wetland functions and capacity.

In addition to this forthcoming inventory (only portions of which are currently available) there is a need to obtain more detailed characterization of potential wetland treatment sites. Greatest attention should be focused on the hydrologic regime to build a more functional picture of particular wetlands [9-3].

MONITORING

In addition to the locations identified in this report, numerous situations exist where pollutants are being discharged to wetland systems. More extensive monitoring of these existing systems should be undertaken for study durations of two to five years [9-4]. Factors requiring consideration include:

(1) Understanding the controlling geology;

(2) Modelling the hydrologic budget, particularly with regard to surface-ground water relationships;

(3) Sediment profiles and analysis;

(4) Intensive water quality analysis, including spatial and temporal differences;

(5) Characterizing the vegetative community and litter components.

In addition to monitoring existing discharges, questions related to the ability of wetlands to recover from pollutant application can be explored through the selection of sites where pollutant discharges have recently been terminated.

Long-term (10 years) monitoring of the most promising wetlands treatment concepts should be carried out in a range of geographical settings to observe equilibrium effects [9-4]. In so doing, it will be incumbent upon the investigators to very carefully document changes in the natural system that could signal future problems.

RESEARCH ISSUES

A number of issues can be identified for future research. These issues, or questions, relate to both an understanding of the fundamental processes involved and to development of management techniques.

Vegetative/Biological System

Most previous studies on vegetative practices for pollutant removal have focused on upland systems, such as crop irrigation, on monocultures in a controlled environment, or upon single plant species. More studies are now needed to address vegetation and other members of the ecological system in the context of wetlands. Specific topics recommended for research include:

(1) Definition of effects on higher level wetland biota as a result of adding pollutants;

(2) Possibility of transfer of heavy metals into the food chain by aquatic plants;

(3) Trace element uptake of various plant species during different parts of the growing season;

(4) Effect of plant type and biomass on degree of treatment;

(5) Nutrient uptake and release by plant litter.

Hydrologic System

The correlation between fluid movement and physical or chemical pollutant removal processes has been extensively studied and is reasonably well understood for controlled environments, such as the primary settling tanks or chlorine contact chambers in wastewater treatment facilities. In wetlands, where many factors will interrelate to produce a unique situation for every site, understanding and control of the system's hydraulics is not as well developed. The following research topics would help reduce this problem:

(1) Development of quantitative information about the hydrodynamics of different wetland types, including natural and polluted water flows;

(2) Classification of the relationship between water movement in a marsn and physical or chemical pollutant removal processes;

(3) Effect of detention time on treatment efficiency for different wetland types.

System Performance

The following research topics would extend previous narrowly focused study results into the more complex interrelationships among pollutant processes in wetlands:

(1) Determination of capacities and functional longevity of natural systems;

(2) Determination of interaction and relative importance of wetland characteristics in achieving wastewater renovation;

(3) Influence of other biological and microbiological factors on wetland performance;

(4) Effect of bottom substrates on pollutant uptake by vegetation and degree of treatment;

(5) Effect of grass/wetland configuration on pollutant removals.

Theory of Wetlands Treatment

Closely related to an understanding of wetlands as a complex treatment system is the development of a theoretical basis for explaining system behavior. Just as the Monod and Michaelis-Menten relationships for organism growth and food usage formed an early basis for sanitary engineering design and operating principles, similar theoretical foundations need to be laid for wetlands utilization. The following topics would help achieve this goal:

(1) Definition of removal kinetics in terms of wetland type, vegetation, detritus, climate and management practices;

(2) Development of an ability to measure and forecast appropriate loading factors and capacities of wetlands.

Wetlands Management

Heretofore, wetlands use for pollutant removal has been, primarily, haphazard or incidental with little control extended over factors that could significantly alter pollutant removal efficiencies or wetland response. In some areas, such as management of wetlands vegetation, far less is known as compared to manipulation of hydraulics, for which some experience can be borrowed from conventional wastewater treatment technology. The following research topics would serve to produce the information necessary for the development of wetlands management procedures:

(1) Development of methods to manage vegetation in wetlands;

(2) Effects of vegetation management, such as harvesting, on nutrient and pollutant uptake;

(3) Development of seasonal flow manipulation techniques for maximum pollutant removal;

(4) Understanding of the effect of water level control on changes in biological pollutant removal and water quality;

(5) Determination of costs and benefits of wetlands treatment of stormwaters and treated municipal wastewaters;

(6) Options or procedures for the reclamation or renovation of wetlands damaged by previous excessive pollutant loadings.

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APPENDIX A

VEGETATION TYPES

The wetland and upland ecosystems are dominated by various characteristic species of vegetation as discussed in Section 2 of this report. These vegetation types are adapted to various environmental and climatic conditions, and population characteristics can vary by geographic distribution. Thus, the same or similar plant species can exhibit narrow tolerances in one area and yet in another climate, such as coastal or desert, the population may withstand more extreme conditions.

Wetland vegetation types can be divided into three categories:

(1) Emergent - rooted in sediments and growing through the water column above water level (Table A-1 and Figure A-1);

(2) Floating - aquatic roots with plant parts partly submerged or fully exposed on the water surface (Table A-2 and Figure A-2);

(3) Submerged - rooted in sediments or aquatic, with all plant parts growing within the water column (Table A-3 and Figure A-3).

In emergent and floating vegetation types, where photosynthetic activity occurs mainly above the water surface, uptake of nutrients, minerals and trace contaminants occurs primarily through the root system. In submerged vegetation, which often has weak root systems, photosynthesis occurs underwater and uptake of compounds can occur directly through stem and leaf tissue as well as through the roots.

TABLE A-1. EMERGENT VEGETATION TYPES, CHARACTERISTICS AND ENVIRONMENTAL TOLERANCES

Vegetation type	Aboveground height	Belowground characteristics	Plant occurrences or tolerances					Comments and references
			Soil	pH (water)	Salinity	Water depth	Air temp*	
Cattails (Typha spp.)	0.7-2.7 m spreading mats and thick stands	Woody rhizomes comprise 45-60% of plant biomass; lives 1½-2 yrs. growth up to 100 cm/yr.	Mud and silt with 25 cm detrital + humic layers; organic content up to 33%.	4.7-10.0 wide range	0.15 ppt (up to 25 ppt some spp.)	0.2-1 m	9-31°C seed germ. at 18-24°C	T. domingensis (coastal + inland growth inhibited at 26 ppt salinity. California population stunted at 12 ppt; T. augustifolia ranges from 0.2-25.5 ppt, optimum growth at <16.2 ppt, cannot tolerate prolonged salinity (>22.5 ppt); T. latifolia does not occur in coastal bays - probably less salt tolerant than T. augustifolia. ((A-6, 7, 8, 9, 12, 16, 17, 45, 50)
Reeds (Phragmites spp.)	3-4 m open stands	Dense rhizomes live 3-6 yrs. - viable to 9 yrs. in dry soils; leafy stolons up to 9 m long.	Clay, sand or silt under swampy conditions; organic content 6-54%	Wide range 2.0-8.5	0-10 ppt optimum; up to 40 ppt some spp.	60-80 cm optimum; ranges from -.3 m to 4 m	11-32°C Seed germ. at 10-30°C.	P. communis of widest distribution and most study; pioneer in early plant succession, but competes poorly; tolerates little water movement; roots can metabolize anaerobically, so reeds can grow in highly reduced muds. (A-1, 7, 8, 9, 10, 11, 36, 37, 47, 48, 49)
Rushes (Juncus spp.) (over 200 spp.)	0.02-1.5 m dense clumps	Creeping rhizomes	Organic soils (no data on content)	No data	0-14 ppt (up to 35 ppt some spp.)	At or just below soil surface.	16-26°C.	Can withstand wave action; found in fresh, brackish & salt marshes; J. balticus tolerates salinities of 16-24 ppt; limitations for coastal spp: J. maritimus 14 ppt; J. balticus and J. gerardii 16-17 ppt; J. maritimus requires <7 ppt for germination. (A-18, 19, 20, 21, 50)
Sedges (Carex spp.)	0.1-1 m in clumps	Creeping rhizomes	Muck or clay with up to 10 cm detrital layer; up to 45% organic content	4.9-7.4	0-.4 ppt (freshwater only)	-.05 to .95 m	15-21°C (range of 14-32°C)	Carex reproduces almost entirely by vegetative propagation. Unlike other emergents, Carex buds form and shoots emerge year-round. Carex competes poorly with other species (A-7, 13, 15, 29, 33, 38, 46, 52, 53)
Sedges - nutsedges (Cyperus spp.)	8-50 cm	Creeping rhizomes, reproduction from tubers and bulbs	Refer to Carex	7.1 optimum, range of 3-8	0-.4 ppt (freshwater only)	At or just above ground surface (-.05 to .3 m)	32°C optimum (range of 16-45°C)	Nutsedge roots enhance denitrifying bacteria activity - decreasing nitrogen availability to other plants; thrives in temperate climates (A-24, 25, 31)

* During growing season.

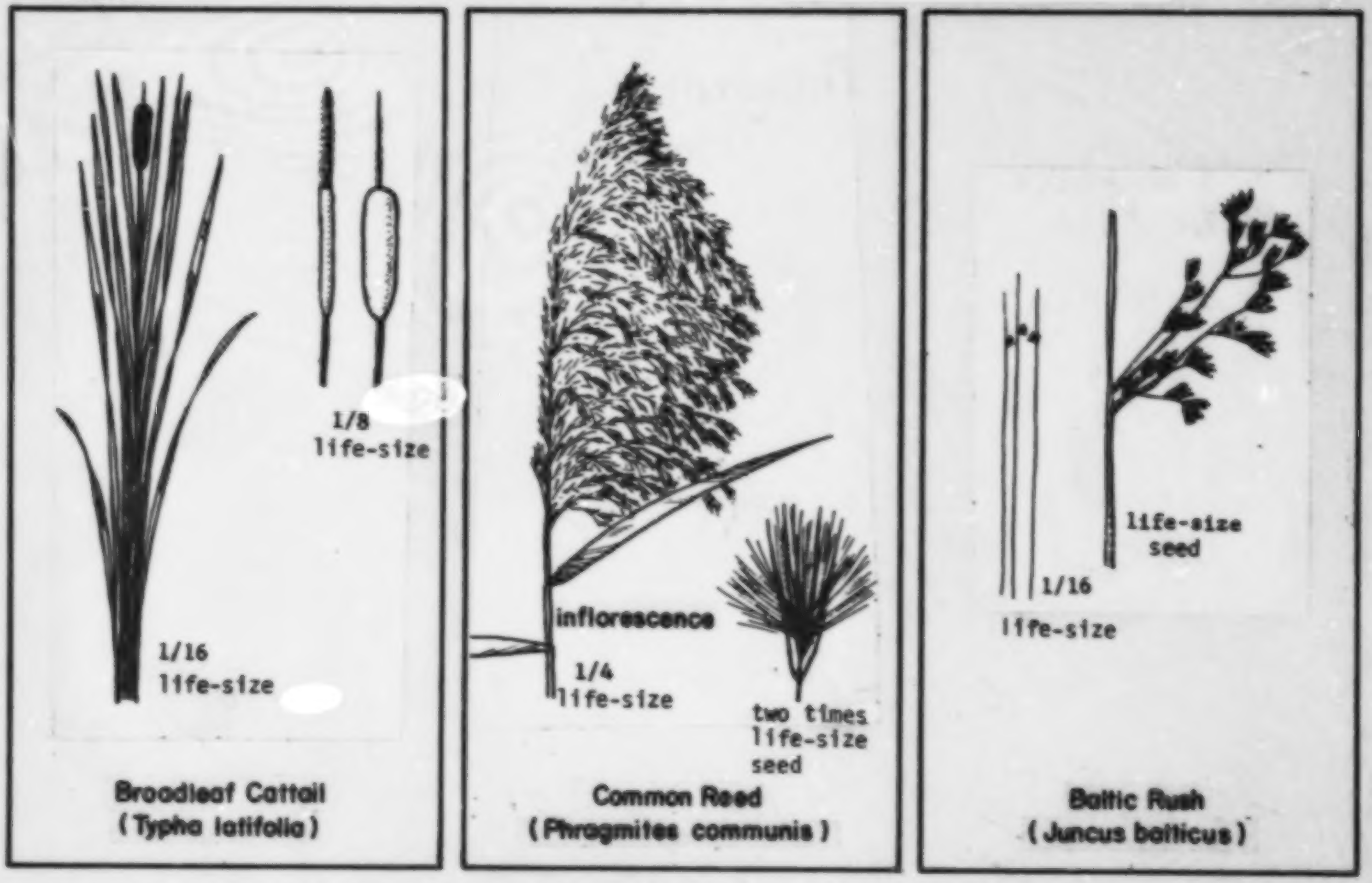
(continued)

TABLE A-1. (continued)

Vegetation type	Aboveground height	Belowground characteristics	Plant occurrences or tolerances					Comments and references
			Soil	pH (water)	Salinity	Water depth	Air temp*	
Sedges - bulrushes (*Scirpus* spp.)	0.3-3 m	Thick, creeping rhizomes	Refer to *Carex*	4-9	4-20 ppt optimum, 0-32 ppt range	Minimum 5-10 cm, maximum depth variable with species	17-28°C, varies with geographical location.	Water level is critical for *Scirpus* success: *S. olneyi* -.75 cm to 6 m; *S. acutus* and *S. heterochaetus* up to 2 m; *S. validus* <1 m; *Scirpus* spp. at Suisun Marsh, Calif. 0.75-25 cm. All *Scirpus* require freshwater for germination; *S. acutus* tolerates salinities to 32 ppt; *S. validus* + *S. heterochaetus* restricted to lower salinities; *S. olneyi* inhibited by >20 ppt; *S. robustus* thrives below 22 ppt. (A-2,3, 8,9,12,39,50,54)
Canarygrasses (*Phalaris* spp.)	30 cm. flower spikes 0.6-2 m	Creeping rhizomes	Silt loam (no data on content)	6.1-7.5	No data, but generally freshwater only	At or just below soil surface	15-25°C (up to 30°C some spp.) Seed germ. at 18-35°C.	Generally an upland grass that tolerates high moisture, high water tables and occasional innundation. *P. arundinacea* (reed canarygrass) is most commonly found and subject to study. (A-4,14,27,28,30,32, 35,51)
Cordgrasses (*Spartina* spp.)	Short forms: 0.3-0.4 m, tall forms: 1.2-2 m	Scaly rhizomes	Sandy substrate	4.7-7.8	9-34 ppt	-.15 to .7 m	12-29°C Seed germ. at 18-35°C.	*S. alterniflora* tolerates salinities to 32 ppt; *S. foliosa* and *S. pagens* only to 18 ppt; most *Spartina* require low salinity (4-8 ppt) for germination. *Spartina* grows best in freshwater, but cannot compete with cattails and rushes; common in salt marshes. (A-5,22,23,26, 34,40,41,42,43,44)
Pickleweed (*Salicornia* spp.)	0.2-0.6 m dense, erect clumps	Thick spreading roots 2-5 mm thick	Mud, clay with low organic content	3.9-8.0	.4-34 ppt (tolerates up to 80 ppt)	-.15 to .3 m	11-32°C	Found in brackish and saltwater marshes; requires saline conditions for growth; can withstand seasonal drying and alkaline conditions. (A-55)

* During growing season

Figure A-1. Emergent Aquatic Vegetation[a]



1/16
life-size
1/8
life-size
inflorescence
1/4
life-size
two times
life-size
seed
1/16
life-size
life-size
seed

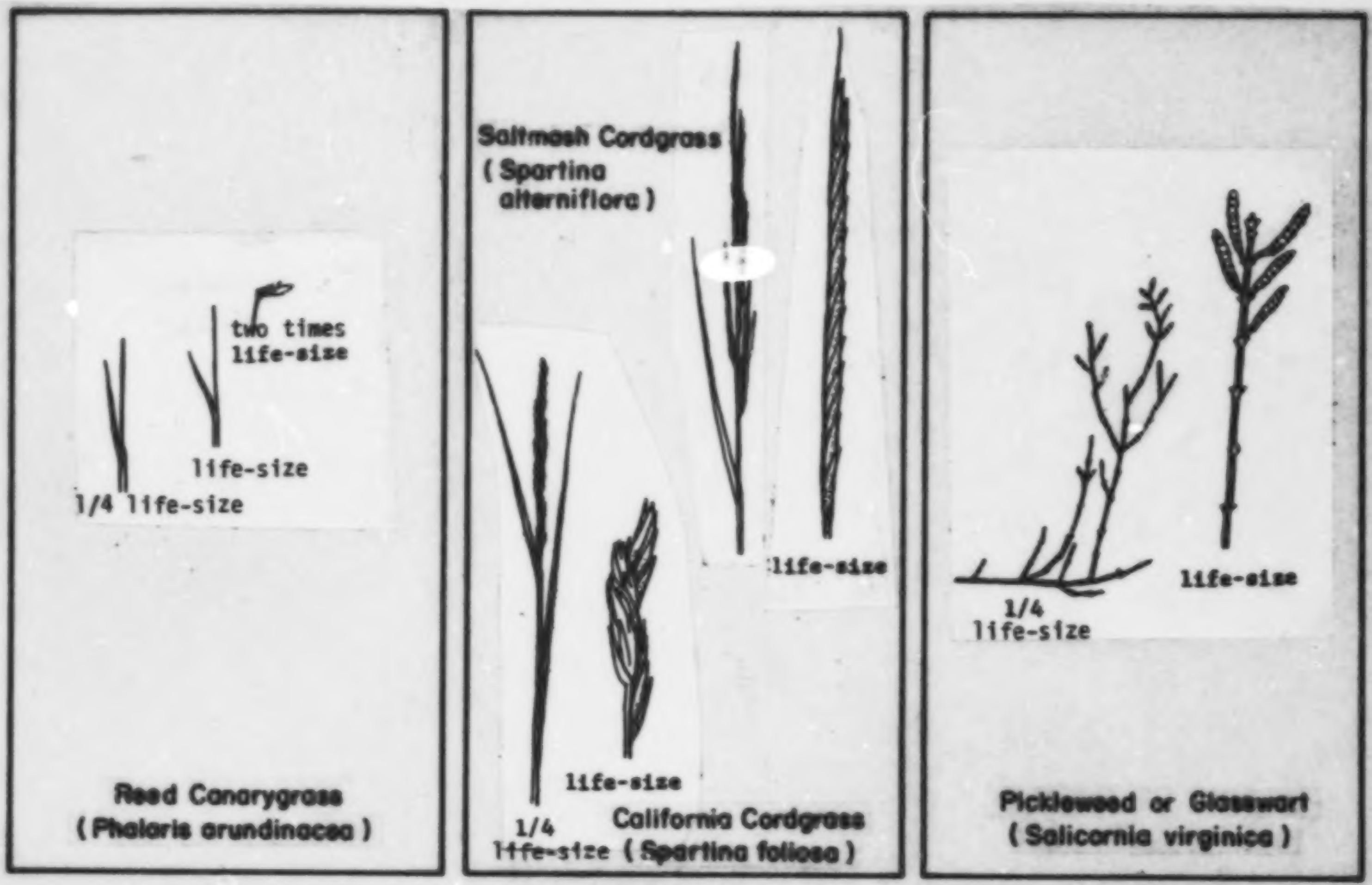
Broadleaf Cattail
(*Typha latifolia*)

Common Reed
(*Phragmites communis*)

Baltic Rush
(*Juncus balticus*)

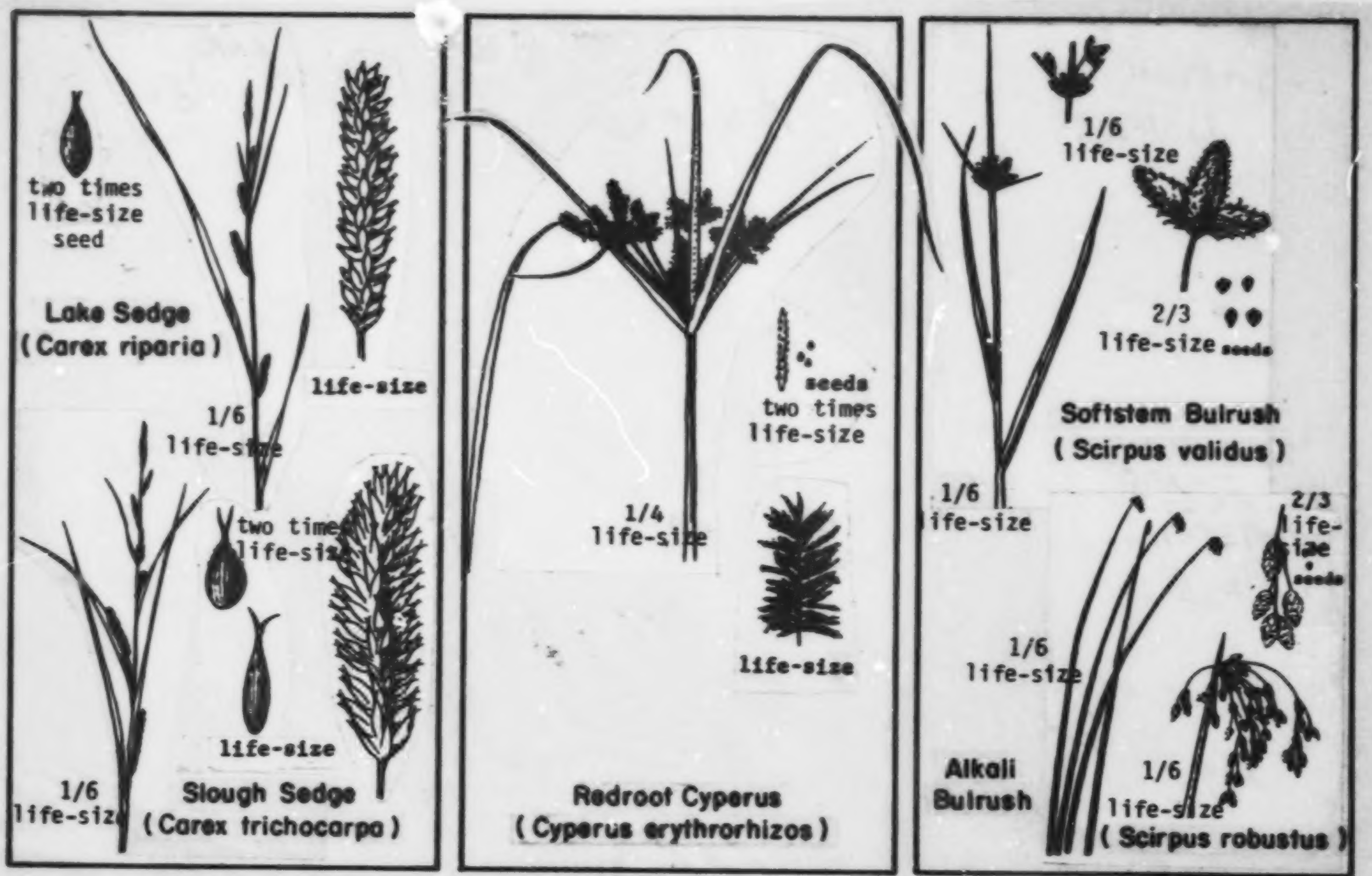
(continued)

Figure A-1. (continued)


two times
life-size
life-size
1/4 life-size
Reed Canarygrass
(Phalaris arundinacea)
Saltmash Cordgrass
(Spartina
alterniflora)
life-size
life-size
1/4
life-size
California Cordgrass
(Spartina foliosa)
life-size
1/4
life-size
Pickleweed or Glasswort
(Salicornia virginica)

(continued)

Figure A-1. (concluded)

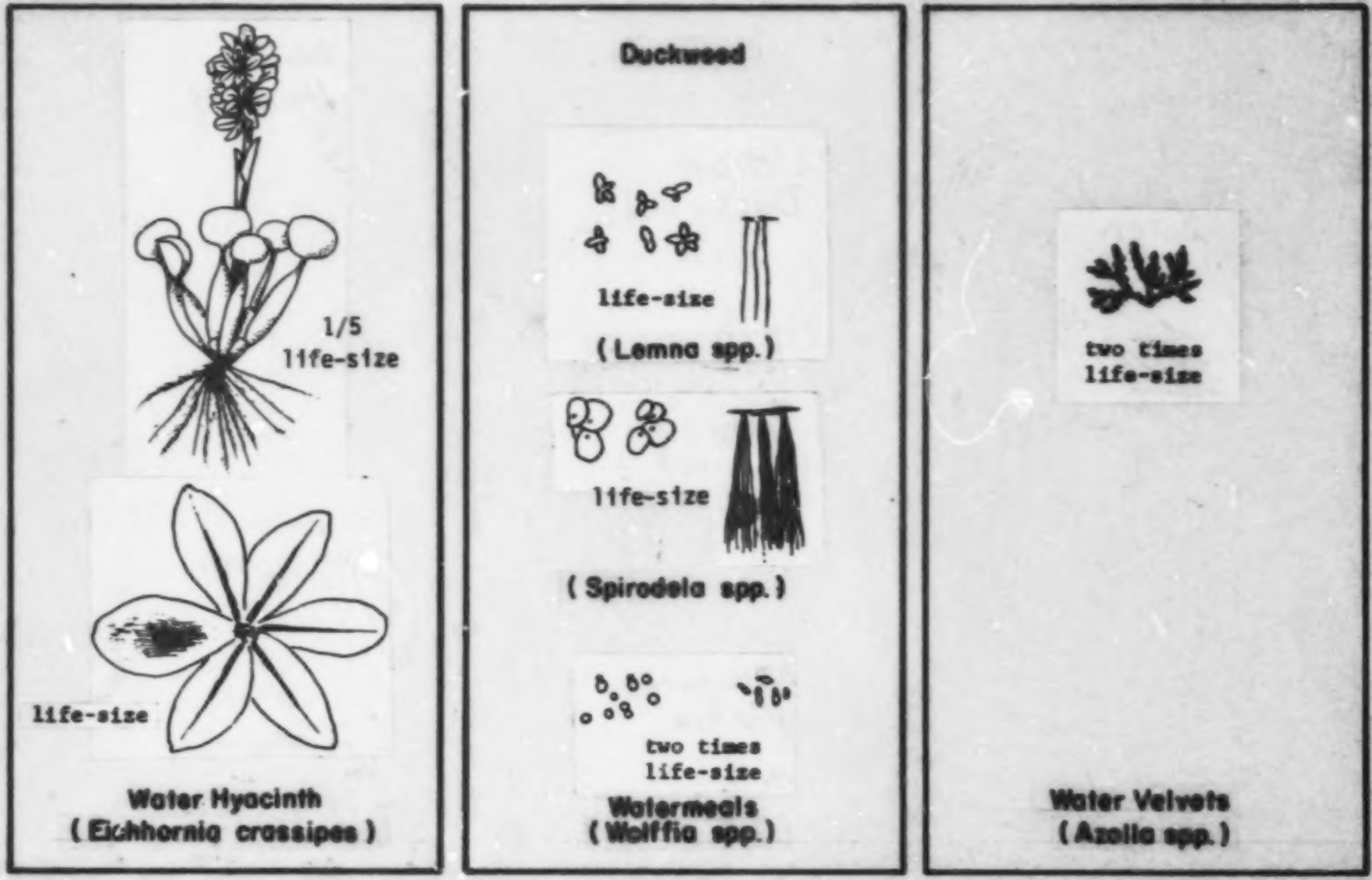

two times
life-size
seed
Lake Sedge
(Carex riparia)
life-size
1/6
life-size
two times
life-size
life-size
1/6
life-size
Slough Sedge
(Carex trichocarpa)
seeds
two times
life-size
1/4
life-size
life-size
Redroot Cyperus
(Cyperus erythrorhizos)
1/6
life-size
2/3
life-size seeds
Softstem Bulrush
(Scirpus validus)
1/6
life-size
2/3
life-size
seeds
1/6
life-size
1/6
life-size
Alkali
Bulrush
(Scirpus robustus)

a. Source: derived from reference A-119

TABLE A-2. FLOATING VEGETATION TYPES, CHARACTERISTICS AND ENVIRONMENTAL TOLERANCES

Vegetation type	Morphology	Plant occurrences or tolerences				Comments and references
		pH	Air temp.	Water temp.	Salinity	
Water hyacinth (Eichhornia crassipes)	Spongy, multi-leaved plant 8 to 127 cm; lavender flowers; under-water rhizomes with long fine roots.	7-optimal; occurs over a wide range	20-30°C optimal; inhibited at 8-15°C; lethal under -5°C.	28-30°C optimal	0-.75 ppt lethal over 2.5 ppt	Tropical plant that will not over-winter in colder temperate climates. Extensive root system serves as a mechanical filter and support structure for bacteria (A-56,60,61,62,63,64, 73,76,77,80)
Duckweed (Lemna spp., Spirodela spp., Wolffia spp., Wolffiella spp.)	Small plants, each with one flat, oval frond 1 cm in length; without roots or with 2 or more threadlike roots on each frond	4.5-7.5 optimal; tolerates wide range	20-30°C optimal.	17.5-30°C optimal; inhibited over 35°C	.75-3.25 ppt optimal; serious effect over 6.5 ppt	Duckweed grows in dense mats in still water, effectively restricting gas transfer and attenuating light; can overwinter in milder temperate climates. Mats of duckweed can raise water temperature up to 12°C above air temperature (A-57,65,66,67,68,69,70, 71,72,73,74,75,78,79,81)
Water fern (Azolla spp.)	Small fernlike plant 1 cm. in diameter with simple roots.	3.5-11	27.5°C optimal; range 5-45°C.	27.5°C optimal	Probably similar to duckweed but can tolerate high salinity.	Nitrogen-fixing blue-green algal symbiont (Anabaena) grows within Azolla, enabling it to colonize nitrogen-deficient waters. Poor growth in open waters with wind/wave action. (A-68,90,73)

Figure A-2. Floating Aquatic Vegetation[a]

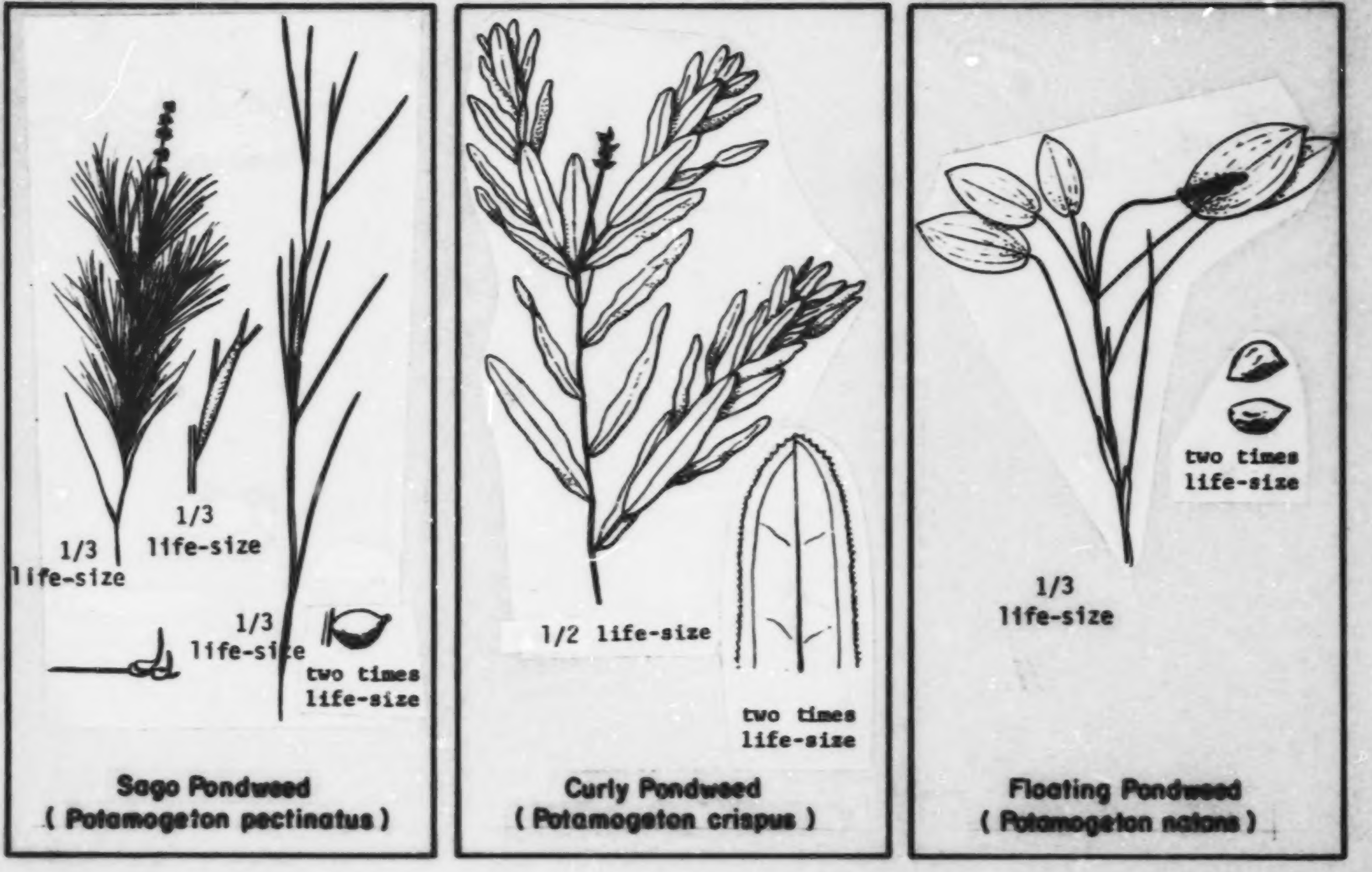

1/5
life-size
life-size
Water Hyacinth
(Eichhornia crassipes)
Duckweed
life-size
(Lemna spp.)
life-size
(Spirodela spp.)
two times
life-size
Watermeals
(Wolffia spp.)
two times
life-size
Water Velvets
(Azolla spp.)

a. Source: derived from reference A-119.

TABLE A-3. SUBMERGED VEGETATION TYPES, CHARACTERISTICS AND ENVIRONMENTAL TOLERANCES

Vegetation type	Morphology	Plant occurrences or tolerances				Comments and references
		pH	Water temp.	Water depth	Salinity	
Pondweed (Potamogeton spp.) 90-130 species	Plants with thread-like to ribbon-like leaves scattered singly on flexible underwater stems; well-developed root system comprises up to 49% of plant mass.	6.3 to 10	10 to 35°C 45° lethal 23-28°C req. for seed germ.	1 to 7 m max. 10 m P. pectinatus opt.=30-46 cm	0 to 15 ppt max. 19 ppt	Reproduces by seeds, winter buds, rhizomes or tubers; tubers are a major waterfowl food; cosmopolitan 90-100 species; P. pectinatus most brackish water and pollutant tolerant; P. crispus, P. pusillus, P. perfoliatus and P. natans also tolerate brackish water and polluted habitats. (A-83,84,89,91,92,93,94,99,100,102,105,106,109,110,112,114,115,117)
Elodea (Elodea canadensis, E. occidentalis, E. densa)	Branched stems 5-15 cm; covered thickly with .5-1.5 cm long leaves; found in groups of 2-6 depending on spp.; well-developed root system.	6.5-10	10 to 25°C 2 to 18°C optimum for P uptake	1 to 7 m max. 12 m	0.2 - 3.6 ppt (E. canadensis) 9.2 - 14.4 ppt (E. nuttallii)	Reproduces by fragmentation; overwinters as winter buds; best adapted for calcareous inland lakes, ponds, slow-moving streams and slightly brackish coastal waters; good water oxygenator; marl deposits are common. (A-82,83,85,89,94,100,106,108,113,116)
Coontail (Ceratophyllum demersum, C. echinatum)	Many-branched bare stalk 1-2 m long; brittle, forked leaves with small thorns; found in clusters of 5-12 from central nodes.	4.0 to 8.7 optimum 7.1 to 8.0	10 to 20°C optimum 18° min. 5.5° (for one species)	No data, probably similar to Elodea	0 to 3.8 ppt	Reproduces by winter buds and fragmentation; requires still or very slow moving water; aquatic birds eat fruits; upright growth in spring; broad floating habit in summer/fall. (A-83,90,98,100,106,107,111,116)
Watermilfoil (Myriophyllum spp.)	Slender stems and few branches; divided leaves grouped in whorls.	5 to 10	0.2 to 30°C optimum 15 to 25°C	1 to 3 m max. 5 m min. 50-80 cm	0 to 15 ppt optimum 0.83-3.33 ppt	Reproduces by fragmentation and/or formation of winter buds; cosmopolitan-45 species. (A-83,86,87,88,89,92,93,94,95,96,97,100,101,103,104,111,118)

Figure A-3. Submerged Aquatic Vegetation[a]

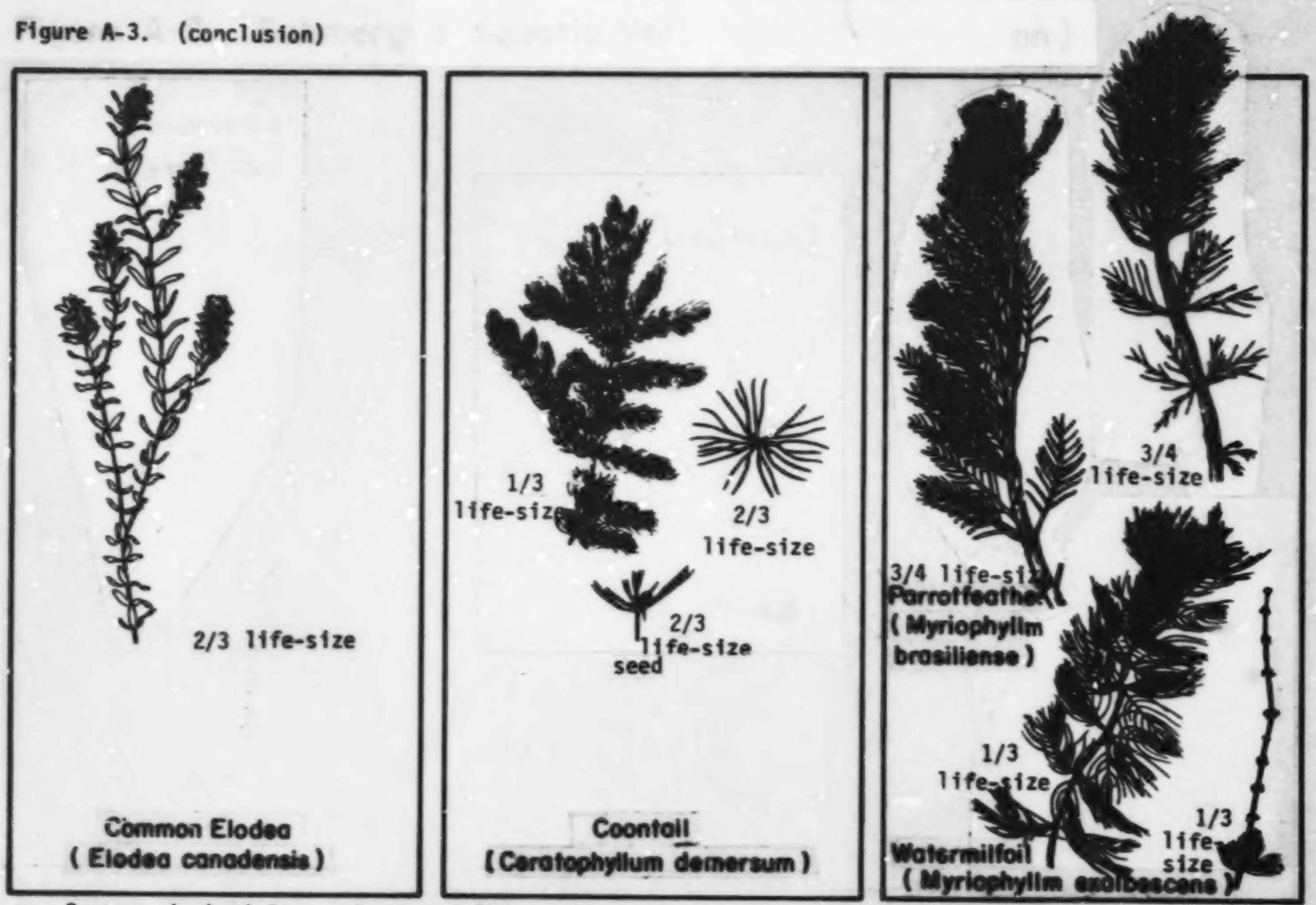

1/3
life-size
1/3
life-size
1/3
life-size
two times
life-size
1/2 life-size
two times
life-size
1/3
life-size
two times
life-size

Sago Pondweed
(Potamogeton pectinatus)

Curly Pondweed
(Potamogeton crispus)

Floating Pondweed
(Potamogeton natans)

Figure A-3. (conclusion)



2/3 life-size
Common Elodea
(Elodea canadensis)
1/3 life-size
2/3 life-size
2/3 life-size
seed
Coontail
(Ceratophyllum demersum)
3/4 life-size
3/4 life-size
Parrotfeather
(Myriophyllim brasiliense)
1/3 life-size
1/3 life size
Watermilfoil
(Myriophyllim exalbescens)

a. Source: derived from reference A-119.

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APPENDIX B

STATE AND LOCAL WETLANDS SURVEY

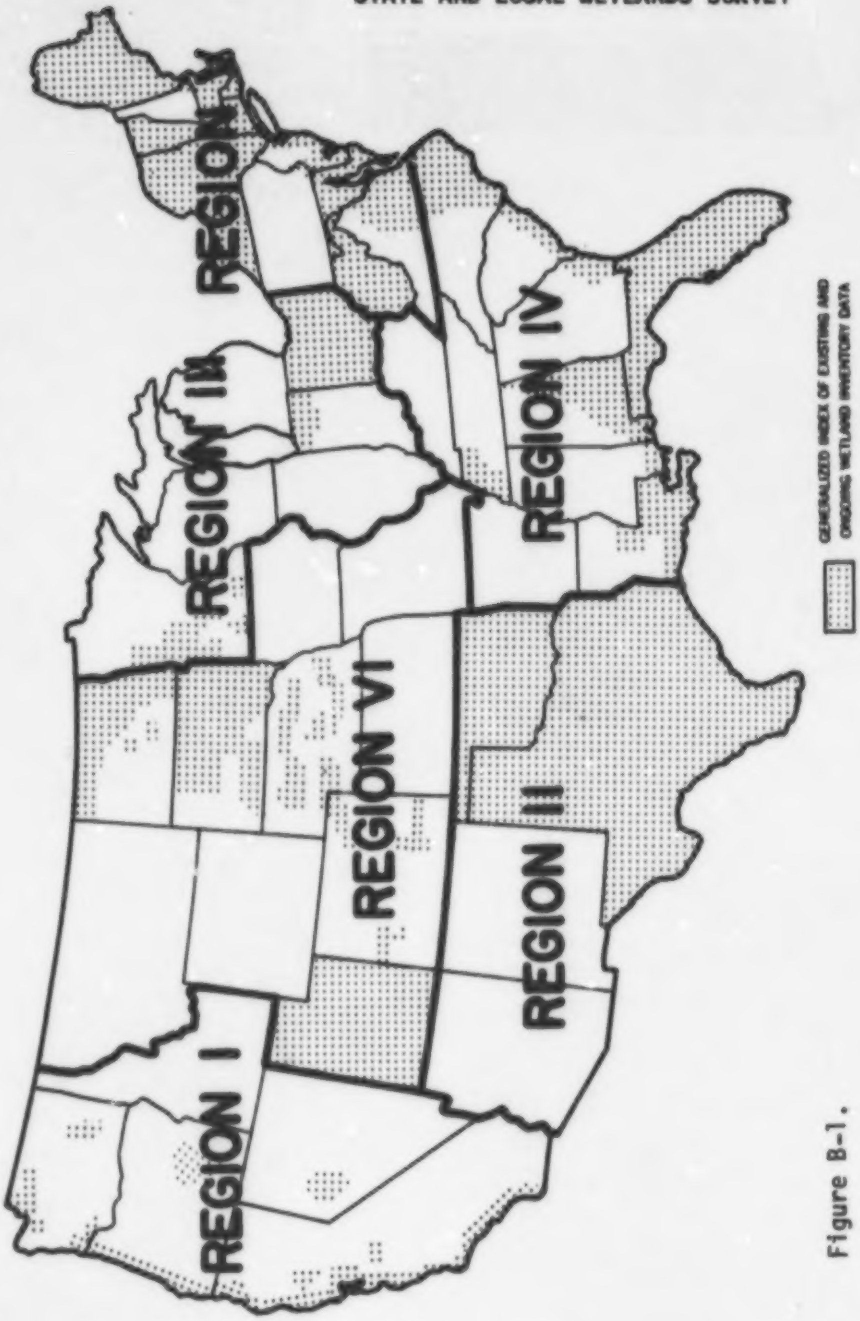
REGION I
REGION II
REGION III
REGION IV
REGION V
REGION VI
▨ GENERALIZED INDEX OF EXISTING AND ONGOING WETLAND INVENTORY DATA

Figure B-1.

EXISTING STATE AND LOCAL WETLANDS SURVEYS

UNITED STATES DEPARTMENT OF THE INTERIOR
Fish and Wildlife Service
Office of Biological Services

ALABAMA

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APPENDIX C

ANNOTATED BIBLIOGRAPHY

WETLAND PROCESSES AND CHARACTERISTICS

C- 1. Bartlett, M.S., L.C. Brown, N.B. Hanes and N.H. Nickerson. Denitrification in Freshwater Wetland Soil. Jour. Environ. Qual 5(4): 460-464, 1979.

This laboratory study demonstrated biological denitrification in freshwater wetland soil and attempted to accurately measure denitrification rates. At least 90 percent of supplemental nitrates were reduced to gaseous nitrous oxide and nitrogen gas, with little or no transfers of nitrate to ammonia or organic nitrogen fractions.

C- 2. Brinson, M.M. Decomposition and Nutrient Exchange of Litter in an Alluvial Swamp Forest. Ecology 58: 601-609, 1977.

Weight loss from cellulose sheets was measured monthly at three sites of a North Carolina swamp forest to determine whether nutrient demand by heterotrophs that decompose forest litter is quantitatively important for the internal recycling and accumulation of nutrients. Rates of loss were significantly different seasonally, and between sites. Both temperature and moisture appeared to be important in controlling decomposition rates. Nutrients appeared to be tightly conserved and recycled, even in swamps subject to flooding during tree dormancy.

C- 3. Engler, R.M., W.H. Patrick, Jr. Nitrate Removal from Floodwater Overlying Flooded Soils and Sediments. Jour. Environ. Qual. 3(4)409-413, 1974.

The floodwater nitrate removal rate of intermittently flooded freshwater swamp soils and continuously flooded saline marsh soils of southern Louisiana was quantitatively characterized in a laboratory study. Of the two areas studied, the marsh area was the more effective sink for nitrate contaminated waters, with an average initial removal rate of 9.15 ppm N/day. The swamp soil had a removal rate of 4.38 ppm N/day. Studies on samples of floodwater separated from the soil showed the active sites of microbial nitrate reduction to be the soil-water interface and within the soil, but not in the floodwater. Additions of organic matter to a mineral soil flooded for rice culture decreased the thickness of the aerobic-anaerobic zone at the soil-water interface and increased the rate of nitrate reduction.

C- 4. Hall, F.R., R.J. Rutherford and G.L. Byers. The Influence of a New England Wetland on Water Quantity and Quality. University of New Hampshire, Durham, N.H., 1972.

Hydrologic, meteorologic and chemical data were collected from a pond/wetland in southeastern New Hampshire to determine its influence on water quantity and quality. Little evidence of systematic or predictable trends was found, since the date showed considerable scatter with time and geographical location. However, the water chemistry showed some evidence of seasonal trends, with total dissolved solids tending to increase from spring into early fall.

C- 5. Isirmiab, N.O. and D.R. Keeney. Contribution of Developed and Natural Marshland Soils to Surface and Subsurface Water Quality. University of Wisconsin, Madison, Wis., 1973.

Preliminary qualitative estimates of the role of natural and developed marshland soils as nitrogen or phosphorus source or sinks were obtained by a limited ground and surface water survey of a marsh adjacent to Lake Wingra, near Madison, Wisconsin, and by laboratory investigations of nitrogen and phosphorus transformations in soil samples from this marsh and from an acid bog in northern Wisconsin. The results indicated that this marsh does not act as a significant nutrient sink, and that though removal of the marsh by draining or filling might result in more nitrogen and phosphorus entering the lake, the presence of the marsh does not appear to lower lake productivity.

C- 6. Lee, G.F., E. Bently and R. Amundson. Effects of Marshes on Water Quality. University of Wisconsin, Madison, Wis., 1969.

The effects of marshes on water quality and the leaching of aquatic plant nutrients from drained marshes were studied at four marsh areas located near Madison, Wisconsin. Marshes were shown to have both beneficial and detrimental effects on water quality. The drainage of a marsh for agriculture-urban purposes was found to cause a significant release of aquatic-plant nutrients, which accelerated the eutrophication of receiving waters.

C- 7. Nicholls, K.H. and H.R. MacCrimmon. Nutrients in Subsurface and Runoff Waters of the Holland Marsh, Ontario. Jour. Environ. Qual. 3(1):31-35, 1974.

The relative surface runoff contributions of nutrients and total electrolytes from cultivated and uncultivated plots of muck soil within the Holland Marsh were studied. Changes in water chemistry and the extent of leaching of nitrogen and phosphorus during the growing season were also recorded. The duration and amount of rainfall, which determined the extent of runoff or percolation into the soil, were shown to be important in determining nitrate and to a lesser extent, soluble reactive phosphorus concentrations in water under the cultivated plot, but not beneath the uncultivated plot.

C- 8. Shih, S.F., A.C. Federico, J.F. Milleson and M. Rosen. Sampling Programs for Evaluating Upland Marsh to Improve Water Quality. Trans. ASAE, pp. 828-833, 1979.

Performance of upland marsh to improve water quality was evaluated using an indirect sampling program which included water chemistry, marsh vegetation, nutrient uptake, soil profile nutrient content, and flow velocity measurement at the Chandler Slough Basin of the Kissimmee River, Florida. The first flush of the wet season, which washes out some of the decaying organic matter, was found to be an important factor affecting nutrient retention in the marsh. The marsh can act as a phosphorus sink if the first flush effect can be controlled by either a retention pond, which reduces the peak flushing flows or, by harvesting the marsh vegetation. Soil sampling showed that the highest nitrogen, phosphorus, and calcium concentration were in the surface layer of soil.

C- 9. Verry, E.S. Streamflow Chemistry and Nutrient Yields from Upland-Peatland Watersheds in Minnesota. Ecology 56:1149-1157, 1975.

Twenty-two water quality parameters were determined over a period of five years for the streamflow from five watersheds with oligotrophic peatlands and one with a minerotrophic peatland. Concentrations of organically derived nutrients were highest in the streamflow from watersheds containing oligotrophic peatlands, while concentrations of nutrients derived from solution of aquifer minerals were higher in streamflow from the watershed containing a minerotrophic peatland. However, flow-weighted concentrations of organically derived nutrients were similar for both watersheds.

WETLANDS UTILIZATION FOR WASTEWATER AND STORMWATER TREATMENT

C-10. Blumer, K. The Use of Wetlands for Treating Wastes -- Wisdom in Diversity? In: Environmental Quality Through Wetlands Utilization. Proceedings from a Symposium Sponsored by the Coordinating Council on the Restoration of the Kissimmee River Valley and Taylor Creek-Nubbin Slough Basin, February 28-March 2, 1978. Tallahassee, Fla ., 1978.

A management policy is needed for the Kissimmee Watershed, Florida to alleviate water quality problems caused by high seasonal nutrient flushes. Research at the Brookhaven National

Laboratory indicates that the utilization of a marsh ecosystem to treat municipal effluent is not as efficient as is a meadow ecosystem in combination with the marsh. Their comparative studies of a meadow/marsh/pond system and a marsh/pond system have shown greater improvements in water quality in the more diverse, meadows/marsh/pond artificial wetland system. It is recommended that this principle be applied to an artificial wetland system for treatment of agricultural runoff in the Kissimmee Watershed.

C-11. Boyt, F.L., S.E. Bayley and J. Zoltek, Jr. Removal of Nutrients from Treated Municipal Wastewater by Wetland Vegetation. Jour. Water Poll. Control Fed. 49(5):789-799, 1977.

The ability of a mixed hardwood swamp to remove nutrients from treated municipal wastewater was studied. A nutrient budget for phosphorus was developed for analysis of the system. A comparison was made between the costs of adding tertiary treatment capabilities to the municipal wastewater facilities and the cost of using natural wetland vegetation for treatment.

C-12. Cederquist, N. Wastewater Reclamation and Reuse Pilot Demonstration Program for the Suisun Marsh. U.S. Department of the Interior, Bureau of Reclamation, Sacramento, California, 1977.

The compatibility of treatment and reuse of municipal and industrial wastewater, and marsh management was evaluated in there phases: high-rate irrigation of pasture to test the use of a living filter to remove nitrogen and phosphorus from wastewater; the use of symbiotic algae ponds for nitrogen removal and for storage of water for flooding the grass marsh ponds; and the cultivation of alkali bulrush and watergrass for waterfowl food.

C-13. Demgen, F.C. and J.W. Nute. Wetlands Creation Using Secondary Treated Wastewater. Proceedings from a Water Reuse Symposium, Vol. I, March 25-30, 1979. Washington D.C., 1979.

The use of treated municipal wastewater to create a wetlands environment for the benefit of wildlife and migratory waterfowl is described, together with a reuse method that combined wastewater and wildlife management for optimum results.

C-14. Dinges, R. Upgrading Stabilization Pond Effluent by Water Hyacinth Culture. Texas Department of Health, Austin, Tx., 1978.

An experimental culture of water hyacinth was used to treat stabilization pond effluent. As a result of the efficient removal of suspended solids and dissolved impurities, water quality improvements were substantial. Effluent from the system was low in nitrogen, low in fecal coliform bacteria and showed large reductions in BOD, total suspended solids and COD. Minerals accumulated in plant material from basin waters were chloride, potassium, phosphorus, arsenic, chromium, mercury, iron, lead, nickel, zinc, copper, magnesium and manganese.

C-15. Dinges, R. Water Hyacinth Culture for Wastewater Treatment. Texas Department of Health Resources, Austin, Tx., 1976.

The capability of water hyacinth to improve the quality of sewage effluent was demonstrated. Literature pertaining to the uptake of nutrients and other minerals by water hyacinth was surveyed. Projects utilizing water hyacinth to treat sewage effluents were reviewed.

C-16. Dinges, W.R. Who Says Treatment Plants Have to be Ugly. Water and Wastes Engr., April, 1976.

The results of a pilot project at the Williamson Creek Wastewater Treatment Plant in Austin, Texas, showed that water hyacinth appears to have the ability to accumulate heavy metals and trace organics. Similar projects indicating the efficiency of water quality improvement by water hyacinth, located in Bay St. Louis, Mississippi; Orange Grove, Mississippi; Washington, D.C; and in various locations of Florida, Louisiana and Colorado, were also reviewed.

C-17. Fetter, C.W., Jr., W.E. Sloey and F.L. Spangler. Use of a Natural Marsh for Wastewater Polishing. Jour. Water Poll. Control Fed. 50(2):290-307, 1978

A stream flowing through Brillion Marsh, Wisconsin, was sampled for water quality at three stations: one located near the point of discharge of sewage effluent, one upstream of the discharge for "marsh background" values, and one downstream of the discharge point for the determination of water quality improvement. The results indicated significant improvements in water quality parameters.

C-18. Grant, R.R. and R. Patrick. Tinicum Marsh as a Water Purifier. Acad. Nat. Sci., Hearings of a Comm. on Merchant Marine Fisheries, pp. 173-191, November 5, 1971,

The Tinicum Marsh, located near Philadelphia, Pa., receives sewage effluent from three municipal treatment plants. A survey of the flora and fauna of the area indicated that the environment has been substantially degraded by the introduction of large quantities of organic wastes. Results of preliminary investigations, however, demonstrated that the marsh, even in its altered condition, can contribute to substantial reductions in BOD, phosphates, nitrates and ammonia and increase the oxygen content of the water.

C-19. Hartland-Rowe, R.C.B. and P.B. Wright. Swamplands for Sewage Effluents, Final Report. Information Canada, Task Force on Northern Oil Development, Report No. 74-4, 1974.

The effects on the local natural swampland ecosystem of released municipal sewage effluent from the town of Hay River, N.W.T., were examined. Water chemistry, bacteria levels, community structure and diversity, nutrient levels in soil and root materials, standing crop and primary production were studied.

C-20. Hickok, E.A., M.C. Hannaman and N.C Wenck. Urban Runoff Treatment Methods, Volume I - Nonstructural Wetland Treatment. EPA-600/2-77-217, Cincinnati, Ohio, December 1977.

A study of the water quality of the Lake Minnetonka Watershed demonstrated the effectiveness of treatment of urban runoff by natural wetlands. The wetland retained 77 percent of all phosphorus and 94 percent of total suspended solids entering the site as a result of various physical, biological and chemical mechanisms.

C-21. Hickok, E.A., E.J. Johnson, N.C. Wenck and M.A. Zagar. Urban Runoff Treatment Methods, Volume II - High-Rate Pressure Filtration. EPA Grant No. S-802535, Cincinnati, Ohio, December 1977.

Stormwater runoff, a nonpoint source of water pollution, substantially contributes to the degradation in quality of the surface waters. High-rate filtration, using three pressure filters, was evaluated as a means of improving the quality of stormwater runoff. In addition, a pilot test was conducted using organic soil filters.

C-22. Houghton Lake Sewer Authority. Concept, Research, Approval: An Effluent Irrigation Project. Consult. Engr. 48(5):77, 1977.

A test program for a marshland irrigation system for the disposal of treatment plant effluents was developed. Financial and environmental considerations were evaluated.

C-23. Kadlec, R.N. and D.L. Tilton. Wastewater Treatment via Wetland Irrigation: Nutrient Dynamics. In: Environmental Quality through Wetlands Utilization, Proceedings from a Symposium Sponsored by the Coordinating Council on the Restoration of the Kissimmee River Valley and Taylor Creek-Nubbin Slough Basin, February 28-March 2, 1978. Tallahassee, Fla., 1978.

Phosphorus and nitrogen were shown to be removed from secondary effluent discharged into the Bellaire wetland treatment system. Average removals of 80 percent to 90 percent for both were indicated. Monthly variability of removal were attributable to patterns of hydrological condition, plant growth and temperature. Decreases in the percentage removal may be caused by low temperatures, low residence times or small effective surface areas.

C-24. Kadlec, R.H., D.L. Tilton and J.A. Kadlec. Nutrient Removal from Secondary Wastewater in a Central Michigan Peatland (Houghton Lake, Michigan). University of Michigan, 1977.

Wetland tertiary treatment of sewage effluent is planned for a peatland at Houghton Lake, Michigan. The background ecology, water chemistry and hydrology of the wetland have been established. Test plots indicated the ability of the wetland to remove nutrients from the sewage. Continued research and monitoring is anticipated following the establishment of the full-scale operation.

C-25. Morris, F.A., M.K. Morris, T.S. Michaud and L.R. Williams. Meadowland Natural Treatment Processes in the Lake Tahoe Basin: A Field Investigation. U.S. EPA, Environmental Monitoring Systems Laboratory, Las Vegas, Nev., 1980.

The effectiveness of natural meadowlands and wetlands in trapping nutrients and sediments from surface runoff was investigated. In systems where the water remained channelized, phosphorus, nitrogen, suspended solids and organic carbon concentrations usually remained unchanged or increased. In systems where the water spread out as sheet flow, total phosphorus, nitrogen, turbidity, TOC, and residue were reduced. The nutrient removal effectiveness of the wetlands approached that of conventional tertiary treatment.

C-26. Mudroch, A. and L.A. Capobianco. Effects of Treated Effluent on a Natural Marsh. Jour. Water Poll. Control Fed. 51(9):2243-2257, 1979.

The effect of a constant supply of effluent from a municipal wastewater treatment facility on the plant communities in a natural marsh area, and the availability and cycling of nutrients and heavy metals in the marsh were studied. Increased quantities of heavy metals were found in sediment samples, but nutrient levels were consistent with those of unpolluted marsh sediments. Plant species growing in the marsh showed varying levels of increase in concentrations of heavy metals and nutrients.

C-27. Ornes, W.H. and K.K. Steward. Effect of Phosphorus and Potassium on Phytoplankton Populations in Field Enclosures. Agricultural Research Service, Fort Lauderdale, Fla., 1973.

Experimental and controlled field enclosures in a marsh environment were treated with weekly applications of phosphorus and potassium. Results indicated a significantly disrupted environment.

C-28. Pirrung, D.E. Feasibility of Utilization of a Wetland for Nutrient Removal from Secondary Municipal Wastewater Treatment Plant Effluent. Proceedings of the 2nd Annual American Water Resources Conference - Wisconsin Chapter, 1977.

The City of Waupan, Wisconsin, was required to upgrade and expand wastewater treatment facilities as a result of strict effluent limits imposed by the Wisconsin Department of Natural Resources. Among the various alternatives considered was the treatment of secondary effluent by an adjacent wetland. The natural treatment method was shown to be cost-effective for certain levels of treatment.

C-29. Roche, W.M. Wastewater Reclamation and Reuse in the Suisun Marsh, California. National Conference on Environmental Engineering Research, Development and Design. University of Washington, Seattle, Wa., 1976.

The Suisun Marsh program was reviewed and application of the results were discussed.

C-30. Small, M.M. Meadow/Marsh Systems as Sewage Treatment Plants. Brookhaven National Laboratory, Upton, N.Y. 1975.

Two natural, closed systems for sewage treatment, a meadow/marsh/pond and a marsh/pond system, were compared. Land requirements and cost comparisons were made.

C-31. University of Michigan, School of Natural Resources and the College of Engineering. Freshwater Wetlands and Sewage Effluent Disposal. Proceedings of a Symposium, May 10-11, 1976, University of Michigan, Ann Arbor, Mi., 1976.

The findings of research from projects pertaining to the utilization of freshwater wetlands for the treatment of secondary effluent at study sites located in Michigan, Wisconsin, New Jersey, Louisiana, Florida, New York and Minnesota were collected. The studies emphasized the hydrology and nutrient balances of the wetlands.

C-32. Small, M.M. Data Report, Marsh/Pond System. Brookhaven National Laboratory, Upton, N.Y., 1976.

The design and operation of a marsh/pond sewage treatment system were described. Observations and data collected from the system during the year were also provided, but no conclusions were drawn.

C-33. Small, M.M. Marsh/Pond Sewage Treatment Plants. Brookhaven National Laboratory, Upton, N.Y., 1976.

A natural wastewater purification system was investigated for cost effectiveness and its capacity for treatment. The size of the marsh required to treat the wastewater for various population sizes was considered.

C-34. Small, M.M. Natural Sewage Recycling Systems. Brookhaven National Laboratory Upton, N.Y., 1977.

Previous experiments indicated that effective treatment of wastewater can be accomplished through the use of a marsh as a filtering system. Further studies indicated that a limited amount of pre-treatment and increased retention times of the wastewater within the marsh can increase the effectiveness of treatment.

C-35. Small, M.M. and C. Wurm. Data Report, Meadow/Marsh System. Brookhaven National Laboratory, Upton, N.Y., 1977.

Thirteen months of operating data for a natural wastewater treatment system were summarized.

C-36. Small, M.M. Freshwater from Sewage on Long Island. Brookhaven National Laboratory, Upton, N.Y., 1967.

A wastewater purification system utilizing natural systems was found to be an effective method of treatment. Wastewater was applied to a meadow, where it percolated through the soil into the groundwater entering a downstream marsh and, flowed finally, into a fish-stocked pond.

C-37. Small, M.M. Artificial Wetlands as Nonpoint Source Wastewater Treatment Systems. In: Environmental Quality Through Wetlands Utilization, Proceedings from a Symposium Sponsored by the Coordinating Council on the Restoration of the Kissimmee River Valley and Taylor Creek-Nubbin Slough Basin, February 28-March 2, 1978. Tallahassee, Fla., 1978.

The use of a meadow/marsh/pond natural system for the treatment of agricultural runoff from individual farms was assessed. The primary concern was the ability of these small systems to remove nitrogen and phosphorus to the extent necessary to make them cost-effective and energy-conservative eutrophication abatement devices for the Kissimmee agricultural region in Central Florida. Design parameters were defined for meadow/marsh/pond systems that would be capable of meeting the nitrogen and phosphorus effluent requirements for discharge.

C-38. Spangler, F.L., C.W. Fetter, Jr. and W.E. Sloey. Phosphorus Accumulation-Discharge Cycles in Marshes. Water Res. Bull. 13(6):1191-1201, 1977.

Changes in the quality of polluted waters flowing through artificial and natural marshes were monitored. Although phosphorus was removed by the sediments and plant parts, it was discharged following the growing season. Recommended methods to prevent the seasonal discharge of phosphorus into surface waters included using the discharge water for irrigation on land, lagooning the water to recycle later, or using conventional treatment methods.

C-39. Steward, K.K. and W.H. Ornes. Assessing the Capability of the Everglades Marsh Environment for Renovating Wastewater. Agricultural Research Service, Fort Lauderdale, Fla., 1973.

Simulated effluents were applied to Everglades sawgrass to determine the feasibility of recycling wastewater through the marshes. The study indicated that insufficient nutrients were assimilated to justify the use of the marsh system to renovate wastewater.

C-40. Sutherland, J.C. and F.B. Bevis. Reuse of Municipal Wastewater by Volunteer Fresh-Water Wetlands. Proceedings from a Water Reuse Symposium, Vol. 1, March 25-30, 1979. Washington D.C., 1979.

The municipal wastewater treatment system at Vermontville, Michigan consists of two facultative stabilization ponds, followed by four diked, surface-[illegible] fields. The Vermontville system was studied [illegible] and evaluate any features that would make similar systems feasible alternatives for economical wastewater treatment for small communities.

C-41. Tilton, D.L. and R.H. Kadlec. The Utilization of a Fresh-Water Wetland for Nutrient Removal from Secondarily Treated Waste Water Effluent. Jour. Environ. Qual. 8(3):328-334, 1979.

Secondarily treated wastewater was applied to a wetland in the northern lower peninsula of Michigan to test the feasibility of utilizing a freshwater wetland for tertiary wastewater treatment. Surface water quality, plant productivity, and nutrient status of plants and soils were measured.

C-42. Tourbier, J. and R.W. Pierson, Jr. (eds). Biological Control of Water Pollution. University of Pennsylvania Press. Philadelphia, Pa., 1976.

This collection of short articles on the feasibility of using higher green plants to purify water was compiled for the International Conference on Biological Water Quality Improvement Alternatives. Biological systems were shown to be potentially effective alternatives or additions to conventional water treatment systems.

C-43. Valiela, I., J.M. Teal and W. Sass. Nutrient Retention in Salt Marsh Plots Experimentally Fertilized with Sewage Sludge. Estuar. Coast. Marine Sci. 1:261-269, 1973.

Salt marsh experimental plots fertilized with commercially available sewage sludge were shown to retain substantial amounts of applied nutrients. The mechanisms by which this occurred, however, could not be clearly defined.

C-44. Loltek, J. and S.E. Bayley. Removal of Nutrients from Treated Municipal Wastewater by Freshwater Marshes. University of Florida, Gainesville, Fla., 1978.

The ability of a central Florida freshwater marsh to assimilate phosphorus and nitrogen from municipal effluent was evaluated. During the initial year of operation, the marsh plots acted as efficient tertiary treatment facilities for the removal of phosphorus and nitrogen.

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C-45. Aulenbach, D.B. Long Term Recharge of Trickling Filter Effluent into Sand. U.S. Environmental Protection Agency, EPA-600/2-79-068, 1979.

The Lake George Village, New York, Sewage Treatment Plant, which applies domestic wastewater that has been subjected to trickling filter purification and secondary sedimentation to a natural delta sand deposit by the rapid infiltration method, was studied.

C-46. Denham-Blair and Affiliates, Inc. and Engineering Enterprises, Inc. Long-Term Effects of Land Application of Domestic Wastewater. U.S. Environmental Protection Agency, EPA-600/2-79-145, 1979

The long-term effects of applying treated domestic wastewater to an infiltration site at Milton, Wisconsin, were studied. Water and soil samples from the site were compared with similar samples from an upstream control area. Differences between sites in mean concentrations of 48 parameters were found to be statistically significant for the effluent applied to the test site as compared to the ground water at the control site. Soil samples from areas surrounding the infiltration lagoon indicated that accumulation of phosphorus, nitrogen and zinc were localized within 150 m of the infiltration lagoon.

C-47. Hinesly, L.D., R.E. Thomas and R.G. Stevens. Environmental Changes from Long-Term Land Applications of Municipal Effluents. Technical Report for the U.S. Environmental Protection Agency, 1978.

Soil and plant samples collected from sewage effluent disposal sites during March 1975 at Bakersfield, California, and during June 1976 at Lubbock, Texas, were analyzed. At both sites, approximately 16 mgd of effluent is applied daily throughout the year on land used for the production of raw and forage crops. Except for changes in phosphorus concentrations, soil chemical properties were not markedly affected by sewage effluent irrigations. Long-term irrigations with sewage effluent caused very little change in the chemical composition of plants grown on the disposal sites.

C-48. Hook, L.E and L.T. Kardos. Nitrate Leaching during Long-Term Spray Irrigation for Treatment of Secondary Sewage Effluent on Woodland Sites. Jour. Environ. Qual. 7(1):30-34, 1978.

Nitrate and total nitrogen were measured at two treated wastewater discharge sites at the Pennsylvania State University Wastewater Renovation Project. The hardwood forest site, which was irrigated year-round with secondary municipal effluent at a rate of 5 cm/week, was ineffective in keeping nitrate concentrations below 10 mg/l. The old field site planted with white spruce was irrigated from April to November each year with 5 cm/week of effluent, and the nitrate concentration at the 120 cm depth rarely exceeded 10 mg/l.

C-49. J. Jenkins, T.F. and C.J. Martel. Pilot Scale Study of Overland Flow Land Treatment in Cold Climates. Prog. Wat. Tech. 11(4-5):207-219, 1979.

The treatment effectiveness of overland flow over a 12-month period of continuous use in a cold temperature climate was investigated at the Cold Regions Research and Engineering Laboratory. Primary and secondary wastewater were applied to separate sections of an overland flow site planted with orchard grass and tall fescue. Removal efficiency of suspended solids remained high throughout the winter, while removal of BOD declined to unacceptable levels at soil temperatures below 4°C. Nitrogen removal declined rapidly below 14°C, and phosphorus removal by overland flow, found to be about 80 percent during the summer months, declined to zero during the winter months.

C-50. Kemp, M.S., D.L. Filip and D.B. George. Overland Flow and Slow Rate Systems to Upgrade Wastewater Lagoon Effluent. Prog. Wat. Tech. 11(4-5):227-256, 1979.

The minimal BOD and suspended solids removal that were attained in this study indicated that overland flow may not be suitable as a tertiary treatment of effluent at this site. Because of low soil permeability, the slow rate system was found to be an excellent tertiary treatment method only if the ground water is protected and no subsurface water collection and discharge is required.

C-51. Tee, C.R. and R.E. Peters. Overland Flow Treatment of a Municipal Lagoon Effluent for Reduction of Nitrogen, Phosphorus, Heavy Metals and Coliforms. U.S. Corps of Engineer Waterways Equipment Station, Vicksburg, Miss., 1979.

Overland flow treatment of municipal facultative lagoon effluent was studied at Utica, Mississippi. Amount of wastewater applied, length of application period, slope of treatment area, crop management, and reduction in BOD, suspended solids, nitrogen, phosphorus, heavy metals and coliforms were evaluated. Overland flow treatment of municipal wastewater was found to produce grass yields on marginal agricultural soils that approached yields obtained on better agricultural soils using normal irrigation water. It was also shown to be a viable method for reduction of nitrogen, phosphorus and heavy metals in municipal wastewater.

C-52. Quin, B.F. Surface Irrigation with Sewage Effluent in New Zealand: A Case Study. Prog. Wat. Tech. 11(4-5):103-126, 1979.

Twenty years of irrigation of pasture with treated sewage effluent was shown to have had a decidedly beneficial effect on the nutrient status of the soil, and on pasture production, which was 50 percent higher than that of nonirrigated land. The fertile soil is no longer capable of removing nutrient from the effluent, however, and nutrients are being lost in the drainage.

C-53. Reynolds, J.H., M.O. Brown, W.F. Campbell, W. Miller and L.R. Anderson. The Long Term Effects of Land application of Wastewater. Prog. Wat. Tech. 11(4-5):283-300, 1979.

A site which had received treated municipal effluent for 20 years and a site which had been irrigated with normal irrigation water were compared over a two-year period. Analysis of water quality, soil characteristics and plant characteristics were made to determine the long-term effects of land applications of wastewater.

C-54. Scott, T.M. and P.M. Fulton. Removal of Pollutants in the Overland Flow (Grass Filtration) System. Prog. Wat. Tech. 11(4-5)301-313, 1979.

An overland flow land treatment system that has been in operation at Melbourne, Australia, since the early 1930's was shown to offer good treatment for BOD, suspended solids, and heavy metals. Nitrogen and/or phosphorus removal are critical, however, were shown to be limited.

C-55. U.S. Environmental Protection Agency. Process Design Manual for Land Treatment of Municipal Wastewater. EPA 625/1-77-008, 1977.

A rational procedure for the design of land treatment systems was presented. Slow rate, rapid infiltration and overland flow processes for the treatment of municipal wastewater were given emphasis. The basic unit operations and unit processes were discussed in detail, and the design concepts and criteria were presented.